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Terrestrial Umbrella – Effects of Eutrophication and Acidification on Terrestrial Ecosystems

Final Report

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Introduction

This report describes the work carried out for the NERC-DEFRA Terrestrial Umbrella (2004 – 2007). Key findings have been synthesized for key topic areas. For individual outputs from Workpackages and Tasks see individual chapters of the report.

Evidence of change

There is a continuing requirement for evidence of both damage and recovery in terrestrial systems. The TU has utilized both survey and experimental approaches to produce new datasets which contribute to this evidence base. Highlights reported here include: (i) Evidence of change in Scottish montane vegetation communities over the last 30-40 years reported which was greatest in areas of enhanced N deposition (with oxidised nitrogen playing the more important role) although N deposition was less important than grazing pressure and climate change (ii) Analysis of the large national monitoring programme, Countryside Survey which indicates a warmer, more lowland climate plus high N deposition seem to track the incursion of nutrient-demanding species into upland Britain, (iii) During the last 50 years, nitrate concentrations in two oligotrophic lakes, Buttermere and Wastwater (Cumbria, UK) have risen significantly, by 70 and 100% respectively. (iv) Synthesis of long term monitoring of foliar and soil solution chemistry in forest stands have demonstrated continued recovery from acidification. (v) Evidence from the long term N addition studies lower plants have identified the most responsive elements within different habitats and relative effects of wet and dry deposition and reduced versus oxidized deposition.

Effect of ammonia

The unique Whim experiment enables the effects of ammonia on a bog system to be studied in detail. Results show negative effects of ammonia are species-specific, with sensitivity related to characteristics that influence the deposition, uptake and assimilation of ammonia, and the consequent effects of ammonia on species' ability to deal with other stress whether abiotic *e.g.* drought and biotic *e.g.* pests and pathogens. These results have been central to a revision of critical level for ammonia.

Indicators of change

An indicator is a marker / proxy measurement which provides evidence for the N status of a site. There are no unique indicators for N status but we have tested a range which when used together indicate that N status has shifted in response to long term N deposition. Using our experiment field sites, we also propose some new indicators which require further testing but could be valuable in the future.

Understanding underlying mechanisms

We have continued to seek evidence for mechanisms underlying changes in vegetation responses. These vary in their importance between habitats but the importance of changes in biogeochemical cycling is highlighted in many. For soils and waters, the importance of soil

type in addition to soil C/N was highlighted together with changes in microbial functioning between soil types and under extreme pollution conditions. A review of peat pH values has been conducted with the objective of providing a robust basis for confirming or amending the critical pH for setting Critical Loads for acidity for organic soils in the UK. The review provides a basis for the re-evaluation of the critical peat pH of 4.4.

Climate change

An examination of the interactive effects of climate change and N deposition were embedded in both survey, experimental, modeling, and critical loads activities within the project. As already discussed, trends in vegetation changes were clearly linked to climate variables in both analysis of the national monitoring programme Countryside Survey (Task 15) and Scottish montane work (Task 9). Experimental evidence highlighted the increased sensitivity of vegetation under climate extremes and also the probably additive effect of some aspects of climate change and N deposition in some sensitive systems.

Modelling

Work has continued to develop the vegetation component of the model chain linked to development of the soils model in the Critical Load and Dynamic Modelling Umbrella. Updated and finalized empirical niche models have been produced for 318 bryophytes and 851 higher plants species. In readiness for GB-level regional predictions of change in plant species habitat suitability, a method has been devised and tested for selecting species pools for each 1 km square. Each species list then reflects those indicators present on a modeled site or within the particular 1km pool that can be modeled using the linked soil and vegetation modules.In recognition of the importance of testing scenarios of biodiversity change driven by multiple drivers, we have continued to develop a second generation of niche models that include climate variables and their interactions with abiotic gradients. These require further development and testing.

The HEATHSOL-UK model was used to assess the long-term impacts of nitrogen deposition on lowland heaths over the period 1850 to 2050 under different management regimes. New historical deposition scenarios were generated for four regions of the UK by CEH Edinburgh, which included spatial variation within a region. Our modeling exercise suggests that historical management, rather than historical deposition, has important consequences for lowland heath response to recent increases in nitrogen deposition.

Rare species

Until now, very little has been known about the likely effects of N deposition on rare species in any UK habitat. In Task 16, we used sand dunes as a model system to test an approach. While many factors contribute to rarity, this study has provided the first data predicting how sensitive some rare species are to changes in the soil bio-chemistry caused by increased N deposition, using dune slacks as a model habitat.

Critical Loads

There have been no significant changes to the national critical loads data sets since February 2004. Consequently part of Task 1 which supports the UK Critical Load National Focal Centre at CEH Monks Wood has been mainly concerned with providing advice on the existing data sets in relation to the development of dynamic models to link soil biogeochemistry to above-ground biodiversity.

A key highlight regarding revision of critical loads has followed the results from the Whim experiment which were central to the setting of a new Critical Level for ammonia of 1 μg m^{-3} (cf. previous value of 8 μg m^{-3}) for ecosystems containing lichens and bryophytes.

A more objective approach to assessing empirical critical loads of nitrogen, using differentiated values within the range recommended for application within UNECE, based on ecosystem characteristics. With respect to climate change, an assessment of changes required for forests has been undertaken and methodology developed. Experimental results suggest the increased supply of N to plants may require increased removal of N though management or a decrease in the N critical load. However, a more complete analysis for a wider range of habitats together with application of models under development is required. Results from the new field surveys are not inconsistent with current critical load ranges. The manipulation experiments also continue to be broadly consistent with these ranges although new evidence from our bog experiment and monitoring work suggests there are still uncertainties and issues concerning the importance of concentration and N form.

Extended Executive Summary

Introduction

This report describes the work carried out for the NERC-DEFRA Terrestrial Umbrella (2004 – 2007). Key findings have been synthesized for key topic areas. For individual outputs from Workpackages and Tasks see individual chapters of the report.

Evidence of change

There is a continuing requirement for evidence of both damage and recovery in terrestrial systems. The TU has utilized both survey and experimental approaches to produce new datasets which contribute to this evidence base. Highlights reported here include:

(i) Evidence of change in Scottish montane communities over the last 30-40 years reported under Task 9. Change was found to be greatest in areas with N deposition above the critical load. Across the whole range of sampled communities, oxidised N deposition was significantly correlated with change in vegetation composition but was not as important as grazing intensity and climate change illustrating the importance of understanding all drivers of change and their interaction. Few changes in soil C:N were observed which matches some experimental evidence and is critical information for parameterising our models

(ii) Analysis of the large national monitoring programme, Countryside Survey carried out under Task 15. Evidence is presented that demonstrates plant species less typical of upland vegetation types and more typical of lowland semi-improved grasslands, have increased in occupancy in Countryside Survey (CS) plots in upland Britain between 1978 and 1998. When variation due to other drivers was covaried out, the probability of at least one nutrient-demanding, lowland mesophyte being present in an upland grassland, heath or bog plot in Countryside Survey data in 1998 was best explained by long-term average annual temperature and total modeled N deposition in 1996. However, the richness of nutrient-demanding, lowland mesophytes in CS plots in 1998 was positively correlated with sheep grazing intensity and cover of intensive Broad Habitat in each 1km square. In conclusion, a warmer, more lowland climate plus high N deposition seem to track the incursion of these nutrient-demanding species into upland Britain but more intense agricultural activity, particularly sheep grazing, is best correlated with a greater richness of these species per square metre, and therefore with the most marked changes in local species composition and vegetation character.

(iii) During the last 50 years, nitrate concentrations in two oligotrophic lakes, Buttermere and Wastwater (Cumbria, UK) have risen significantly, by 70 and 100% respectively. This is not related to patterns of fertilization suggesting N deposition is the major controlling factor.

(iv) Synthesis of long term monitoring of foliar and soil solution chemistry in forest stands within Task 14. Key findings include the confirmation of recovery of soil solution pH from high pollution loading at the Ladybower site in the English Midlands and, in addition, recovery is becoming apparent at another acid site in Wales – Llyn Brianne. A downward trend in soil solution sulphate concentration has been observed at most sites. Foliar sulphur content at all sites has also decreased such that at Rannoch, one of the pristine Scottish sites, to a level classed

as deficient. Foliar aluminium levels have continued to fall at most of the conifer sites, suggesting a general recovery from acidifying pollutant loading.

(v) Evidence from the long term N addition studies are reported under Tasks 3,4, 5,6 and 7. In summary, lower plants such as mosses and lichens are found to be the most responsive group of species across all studied habitats; there is evidence of decline in cover and/or species richness or a change in species composition. In grasslands, there is evidence that nitrogen deposition affects forb negatively with reduced flowering whilst grasses increase. Gaseous NH_3, but not wet oxidised or reduced deposition, changed keystone species composition in a bog; N deposition also decreased flowering of sensitive grassland forb species.

Effect of ammonia

The unique Whim experiment in Task 7 enables the effects of ammonia on a bog system to be studied in detail. Results show negative effects of ammonia are species-specific, with sensitivity related to characteristics that influence the deposition, uptake and assimilation of ammonia, and the consequent effects of ammonia on species' ability to deal with other stress whether abiotic *e.g.* drought and biotic *e.g.* pests and pathogens. *Cladonia portentosa* is more sensitive than heather (*Calluna*) which is more sensitive than bog moss, *Sphagnum capillifoliu.* Long-term exposure to ammonia causes plants to be damaged by lower ammonia concentrations *ie.* the damaging effect over time is cumulative. These results have been central to a revision of critical level for ammonia.

Indicators of change

An indicator is a marker / proxy measurement which provides evidence for the N status of a site. There are no unique indicators for N status but some have been identified which when used together indicate that N status has shifted in response to long term N deposition. These have been tested using our combined resources of the long term experimental field sites (Tasks 3,4, 5,6,7) and survey activities (Tasks 9, 10, 14 & 15).

(i) Foliar N is a better indicator of N deposition than of N status (that is appropriate for a particular habitat). In experimental N manipulations, foliar N generally increased in response to N additions, by proportions that were species specific and relatively greatest in the lower plant species. Reduced N usually caused proportionally greater increases than oxidised N with ammonia far more effective at elevating foliar N concentrations and causing biochemical changes, which may be evidence of phytotoxicity, than the equivalent wet deposited N load. However, in regional surveys and long-term monitoring, which involve many other drivers, the relationship between foliar N with N deposition is often complex. Foliar N:P was found not to be a good indicator.

(ii) Litter extractable ammonium and inorganic-N leaching from soil, show a strong relationship with N deposition across a range of experimental manipulations and field surveys, and are good indicators of appropriate N status for a particular habitat. Net nitrification in experiments showed a strong response to N deposition, and its use as as an indicator is supported by other experimental and survey work. C: N was found to have a complex relationship with N deposition.

(iii) Changes in species richness, abundance and diversity are generally not useful as indicators. However, presence/absence/frequency of specific species could be used as an indicator. This would require more detailed analysis of results from different experiments to identify consistent responses.

(iv) Indicators of potential value that have been observed to be useful in one of more sites but require more investigation including soil and litter enymes (phenol oxidase in heathlands has already been tested), soil extractable ammonium, PME activity in bryophytes, soils and litter, and microbial nitrate immobilisation, metabalomics, lichen epiphytic species assemblages and plant reflectance.

Understanding underlying mechanisms

(i) Evidence for mechanisms underlying changes in vegetation responses is derived primarily from field manipulation experiments in Tasks 3,4,5,6 and 7. There is evidence of increased sensitivity to winter damage, summer droughts, and pest and pathogen attack, in *Calluna* and bryophytes. N deposition alters phenology, accelerates developmental cycles, and increases growth and litter production in heathlands and bogs. In acid and calcareous grasslands, there is evidence of effects in reducing soil pH, base cation status, and phosphorus levels, and in reduced mycorrhizal infection of sensitive species. All of the experimental sites show continued immobilisation of N inputs (30 – 95%), and there is evidence of change in microbial communities, although the implications of these for biogeochemical cycling needs further investigation.

(i) For soils and waters, work was carried out in Tasks 11, 12 and 13 to elucidate control mechanisms. In one study in the Lake District, combining observed nitrate leaching rates with estimates of other N losses suggests that the soils are currently accumulating 30–40 kgN/ha/yr. Nitrate leaching was found not to be simply related to soil C:N ratio which currently underpins much of the modeling work highlighting the need to understand underlying at a more detailed level. This more detailed study of processes within the soil 'black box' was attempted using a isotope approach in wetland soils. These habitats were selected as they are at the critical interface controlling N release to surface waters. Results indicated that microbial immobilisation of N was greatest in the more aerobic nutrient poorer (but higher C/N) wetlands located at the ombrotrophic and mesotrophic sites. Conversion of nitrate to ammonium by microbes was identified as a potentially important route for NO_3 consumption in waterlogged mineratrophic peat but is currently not represented in models. In a heavily N & S polluted organic soils in the Peak District, ammonium is not nitrified and neither ammonium or nitrate are used microbially and leak to streams.

(ii) A review of peat pH values has been conducted with the objective of providing a robust basis for confirming or amending the critical pH for setting Critical Loads for acidity for organic soils in the UK. The review provides a basis for the re-evaluation of the critical peat pH of 4.4.

Climate change

The effects of climate are embedded in both survey (Task 9 and 15), experimental (Task 8), modelling (Task 17) and critical loads (Task 8 and 14) activities within the project. As already discussed, trends in vegetation changes were clearly linked to climate variables in both the national monitoring programme Countryside Survey (Task 15) and Scottish montane work (Task 9). This highlights the need to understand these changes will affect sensitivity of ecosystems to N deposition and thus their critical loads.

Our experimental work in Task 8 reports on finding from manipulations in upland ecosystems which warm the system by 1^oC or simulates modest summer drought. These

climate manipulations had a modest effect on N and pH concentrations in soil water relative to the effect of N input. However, there is evidence of increased N supply promoting vegetative growth which can is a key driver of changes due to N deposition suggesting a possible additive effect of climate change and nitrogen on this ecosystem.

Modelling

Work has continued to develop the vegetation component of the model chain linked to development of the soils model in the Critical Load and Dynamic Modelling Umbrella. Updated and finalized empirical niche models have been produced for 318 bryophytes and 851 higher plants species. A new set of calibration equations were produced using a multiple regression approach. These achieved a significant improvement in r-sqrd value when explaining mean Ellenberg indices. While these improved functions still reduce the accuracy of niche models when used to translate soil conditions into species composition they are a vital link between dynamic soil model outputs and GBMOVE niche models.

Predictions of soil pH and C:N ratio for Countryside Survey field plots in 1978 and 1998 were generated using the VSD dynamic soil model. Comparisons with observed soil data showed a lack of fit and emphasize the pressing need for an analysis of the sources of uncertainty involved. It is vital, for example, to understand how much uncertainty is being contributed by modeled N and S deposition errors, unrepresentative soil chemistry data, unmeasured drivers of soil and vegetation change or too crude a representation of soil dynamics within VSD in future work. More accurate predictions of soil pH and C:N are likely when the more sophisticated yet more parameter-rich MAGIC model is applied to British 1 km squares and the tests against observed data repeated.

In readiness for GB-level regional predictions of change in plant species habitat suitability, a method has been devised and tested for selecting species pools for each 1 km square. Each species list then reflects those indicators present on a modeled site or within the particular 1km pool that can be modeled using the linked soil and vegetation modules.In recognition of the importance of testing scenarios of biodiversity change driven by multiple drivers, we have continued to develop a second generation of niche models that include climate variables and their interactions with abiotic gradients. These require further development and testing.

The HEATHSOL-UK model was used to assess the long-term impacts of nitrogen deposition on lowland heaths over the period 1850 to 2050 under different management regimes. New historical deposition scenarios were generated for four regions of the UK by CEH Edinburgh, which included spatial variation within a region. Our modeling exercise suggests that historical management, rather than historical deposition, has important consequences for lowland heath response to recent increases in nitrogen deposition. We predict that rates of deposition prior to 1950 did not lead to significant increases in nitrogen availability, except in regions where maximum rates of deposition occurred. In contrast, long-term impacts of grazing, cutting and burning on the nitrogen budget of a site were predicted to be more significant in terms of sensitivity of heathland status to increased nitrogen deposition (Figure 14). However, this factor is not considered in most current assessments of the impacts of nitrogen deposition. Our findings indicate that caution is needed in interpretation of studies which relate current nitrogen deposition to recent ecological change and to biological indicators of possible change, as variation in historical managements between sites may make the ‘signal’ of nitrogen deposition difficult to detect. The findings relate only to lowland

heath, and some consideration needs to be given to their relevance in other habitats for which historical management practices may have differed greatly from those in the more recent past

Rare species

Until now, very little has been known about the likely effects of N deposition on rare species in any UK habitat. In Task 16, we used sand dunes as a model system as it is a Priority BAP habitat containing over 70 red data book, nationally rare or scarce species, listed in Annex 2 of the Habitats Directive, as well as other species of conservation importance. Many of the rarer species occur in early successional habitats and are highly susceptible to N-deposition driven soil and vegetation change. While many factors contribute to rarity, this study has provided the first data predicting how sensitive some rare species are to changes in the soil bio-chemistry caused by increased N deposition, using dune slacks as a model habitat. Field survey data at 12 sites combined with a simplified N accumulation model and a species-environment model produced a ranking of the sensitivity of selected species to changes in nitrogen deposition.

Critical Loads

There have been no significant changes to the national critical loads data sets since February 2004. Consequently part of Task 1 which supports the UK Critical Load National Focal Centre at CEH Monks Wood has been mainly concerned with providing advice on the existing data sets in relation to the development of dynamic models to link soil biogeochemistry to above-ground biodiversity. However, two additional pieces of work were undertaken concerning uncertainty and use of a index to predict catchment weathering needed in the steady state mass balance approach for calculating critical loads.

A key highlight regarding revision of critical loads has followed the results from the Whim experiment in Task 7 which were central to the setting of a new Critical Level for ammonia of 1 $\mu g\ m^{-3}$ (cf. previous value of 8 $\mu g\ m^{-3}$) for ecosystems containing lichens and bryophytes. This has now been accepted by the relevant parties within the UNECE and is awaiting formal ratification in December 2007. We suggest, the use of these ammonia concentrations rather than equivalent N deposition may be a more useful tool for environmental regulators to operate with, since concentrations can be relatively easily and cheaply monitored, and the calculation of equivalent N deposition, from concentrations leads to large uncertainties.

Results from the new field surveys are not inconsistent with current critical load ranges. The manipulation experiments also continue to be broadly consistent with these ranges although new evidence from our bog experiment and monitoring work suggests there are still uncertainties and issues concerning the importance of concentration and N form.

Work in Task 1 was carried out to develop a more objective approach to assessing empirical critical loads of nitrogen, using differentiated values within the range recommended for application within UNECE, based on ecosystem characteristics. The approach that has been developed, after active consultation with potential users, provides a decision support matrix (DSM) based on a combination of data from the National Vegetation Classification (NVC), site-specific information where appropriate, and endorsement theory to assess the strength of evidence supporting a particular recommended critical load value.

With respect to climate change, Task 14 examined the effects of predicted changes to forest productivity as a result of climate change. Changes in base cations and nitrogen uptake term in the Critical Loads calculations have been evaluated. The evaluation includes also the regional expression of the climate change impacts and effect on the Critical loads exceedance statistics. Methodology for incorporating regional variation in productivity on the growth uptake term is developed, together with an approach to represent the impacts of climate change. Within Tasks 7 and 8, impacts of climate change on critical loads for Calluna moorland and bogs are also discussed. For empirical critical loads, results suggest climate extremes can increase sensitivity to N deposition in bog systems whilst warming increased pathogen attack in heather moorland also associated with N deposition. In terms of the steady state mass balance derived critical load, our results suggest that increased supply of N to plants may require increased removal of N though management or a decrease in the N critical load. However, a more complete analysis for a wider range of habitats together with application of models under development is required.

Work Package 1:
Refinement of Critical Loads

Task 1:
Update and refinement of critical loads

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Task 1 - Update and refinement of critical loads

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1. Summary

There have been no significant changes to the national critical loads data sets since February 2004. Consequently this task has been mainly concerned with providing advice on the existing data sets in relation to the development of dynamic models to link soil biogeochemistry to above-ground biodiversity. However, two additional pieces of work were undertaken. These were:

- An evaluation of uncertainty in the parameter values applied to the steady-state models used to calculate critical loads of nutrient nitrogen and acidity for UK woodland habitats, including a comparison with European estimates. The results showed wide ranges in uncertainty resulting from methodological differences across Europe and from scaling factors. There is a need for coordinated effort to agree upon uncertainty ranges for critical load input data so that a consistent approach is employed across Europe. The results of this task were also used in the evaluation of uncertainties in critical loads at the site-specific, regional and national scales undertaken for the Environment Agency (Skeffington et al., 2007).

- An evaluation of the sodium dominance index (NaDI) model for predicting catchment mineral weathering rates. This used data from applications of the MAGIC model in five acid sensitive regions of the UK. Overall, the NaDI model did not perform as well as a regression model based on the simple concept of the sum of the excess base cation concentrations in the water. In practical terms, the NaDI model is simpler to apply, only requiring the measurement of three base cations in a stream water sample whereas an additional measurement of chloride ion is required to calculate excess base cation concentrations.

2. Policy Relevance

This task provides technical input to the UK National Focal Centre (UKNFC) in support of their mapping activities and provision of data to the Coordination Centre for Effects (CCE). The work takes the form of contributing to the assessment and development of critical load models, provision and review of data and assessment of uncertainties.

3. Objectives

To refine critical loads for the main terrestrial ecosystems and make assessments of critical loads exceedance at the national scale

4. Methods and results

4.1 Support to UKNFC

The national steady-state critical loads data for acidity and nutrient nitrogen were significantly revised and updated in 2003, with further minor updates in 2004. Currently there are no plans to further revise and update these data sets. Keeping the data 'fixed' will help in the current reviews of the NECD, the Gothenburg protocol and the Defra Air Quality Strategy, where the focus is on predicted changes in emissions and deposition.

The CCE issued a voluntary call for data in October 2005 (for delivery January 2006). However, as the UK submitted revised data to the CCE in February 2004 and dynamic modeling outputs in February 2005, and as no further changes had been made since that time, no further data were submitted. The CCE issued a further voluntary call for data in Autumn 2006 (for delivery February/March 2007); in response to this call the NFC provided empirical critical loads of nutrient nitrogen for UK Special Areas of Conservation and dynamic modelling outputs (acidity) for terrestrial habitats (Evans et al, 2007).

A watching brief on further developments in critical loads has been maintained by attendance at relevant national and international meetings such as TU project meetings and "Acid Rain 2005".

4.2 Uncertainty in Steady-State Models

Uncertainties in the data used to calculate steady-state acidity and nutrient nitrogen critical loads for woodland habitats in the UK were examined. The models use data from a wide range of sources and the accuracy of the critical loads are influenced by the accuracy of the input data sets. The work was led by the UK NFC (Elizabeth Heywood and Jane Hall) and aimed to quantify the uncertainties in the data used and to make comparison with values calculated by other European countries. The work identified the maximum ranges of potential values and probability distributions for all input parameters for the calculation of critical loads in the UK, calling upon data sources collated from activities of the Terrestrial Umbrella project. These values were compared with the uncertainties and probability distributions arising from other studies in Europe, notably Austria, Finland, Germany and the Netherlands (CCE work on the European background database). In addition the uncertainties in parameters from three UK site-specific studies (Alice Holt, Liphook, Plynlimon) were also compared with the other UK data and European data. Each parameter was examined individually to identify sources of uncertainty in the data and to compare the methods for estimating uncertainty between the different studies. For example, Table 1 below (extract from Heywood et al, 2006) summarises the findings on uncertainty in the nitrogen uptake term (ie, the amount of nitrogen removed by harvesting or removal of trees) for woodland habitats.

Table 1. Uncertainty in nitrogen uptake for woodland habitats

Country/site	Ecosystem	Uncertainty range	Probability distribution
UK	Coniferous	CV = 27%	Normal
	Deciduous	CV = 7%	Normal
Liphook (UK)	Coniferous	± 50%	Uniform
Austria	Coniferous & deciduous	± 20%	Uniform
Finland	Birch	CV = 37%	Triangular
	Spruce	CV = 55%	
	Pine	CV = 44%	
Germany	Not specified	± 15%	Uniform
Netherlands	Forest soils	CV = 50%	Normal
Sweden	Coniferous & deciduous	± 50%	Triangular

For the UK the coefficients of variation (CV) are derived from the data for the 10 Level II Intensive Forest Health monitoring sites. For the Liphook site, the uncertainty range was estimated from site measurements. Germany used error propagation techniques to determine uncertainty, and Finland used a similar approach. The studies for Austria and Sweden were based on expert judgment.

As can be seen from Table 1 the analysis of uncertainty revealed a wide range in the estimates of uncertainty in model parameters and in the methods used to derive them. Possible reasons for these differences were the range of methods used to estimate uncertainty, the scale at which the uncertainty was assessed and underlying methodological differences used to estimate parameter values.

4.3 Sodium dominance index

This piece of work made a regionalised assessment of the sodium dominance index method proposed by White et al. (1999) as a technique for estimating catchment mineral weathering rates. Mineral weathering is one of the fundamental processes determining soil and freshwater susceptibility to acidification. Unfortunately, weathering rates are extremely difficult to measure and a modelling approach is often required. This may vary in sophistication from simple, semi-quantitative map based assessments to steady state and dynamic, processed based models. White et al. (1999) have proposed that in areas with a strong maritime influence, such as the UK, the sodium dominance index (NaDI), the ratio of the concentration of Na^+ to $\Sigma Na^+ + Ca^{2+} + Mg^{2+}$ in a stream water sample will provide a quantitative index of the catchment weathering rate upstream of the measurement point. The model was originally tested by the authors in a number of Scottish catchments with a wide range of bedrock geology using weathering rates calculated by the MAGIC model to calibrate the NaDI. Here, data from 293 sites in five acid sensitive regions of the UK have been used to further evaluate the performance of the model.

In the regional assessment of the NaDI model, the data set for each region was split randomly into two halves. One half was used to construct simple linear regression models of a) MAGIC weathering rate vs. NaDI and b) MAGIC weathering rate vs. ΣxsBC; the sum of $xsNa^+ + xsCa^{2+} + xsMg^{2+}$. The second half of each data set was used to test the predictive capabilities of the regression models.

Table 2 shows the r^2 values for the model calibrations and for the relationships between the weathering rates predicted by the regression models and MAGIC weathering rates. The model using ΣxsBC invariably gave higher r^2 values for both calibration and testing. For three regions, the means of the residuals (regression model prediction minus MAGIC weathering rate) were lower for the ΣxsBC compared to the NaDI model. The opposite was true for the Cairngorms and Wales. For all regions, the standard deviation of the residuals was higher for the NaDI model, implying greater variability in the predictions. For the NaDI model, residuals were negatively correlated with MAGIC weathering rates for all regions except the Cairngorms. For the ΣxsBC model, the residuals were randomly distributed for the Cairngorms and Galloway, positively correlated with MAGIC weathering rate for the Lake District and negatively correlated for the Pennines and Wales.

Table 2. Values of r^2 for the calibration and testing of the NaDI and ΣxsBC regression models

	Model calibrations			Model testing		
Region	No. of sites	NaDI (r^2)	ΣxsBC (r^2)	No. of sites	NaDI (r^2)	ΣxsBC (r^2)
Cairngorms	19	0.684	0.928	19	0.553	0.926
Galloway	28	0.694	0.886	27	0.670	0.793
Lake District	25	0.754	0.906	24	0.824	0.951
Pennines	27	0.861	0.889	29	0.493	0.802
Wales	48	0.705	0.822	47	0.573	0.659

Overall, the NaDI model did not perform as well as the regression model based on the simple concept of ΣxsBC concentrations.

4.4 Collaboration with Dynamic Modelling and Freshwater Umbrella

The current focus of critical loads activities in the UK and Europe is on nutrient nitrogen and the development of critical loads for nitrogen in relation to biodiversity. Dynamic models (eg MAGIC-GBMOVE) are being developed to link soil biogeochemistry to plant species diversity to inform this process. The national critical loads data and the data sets used to derive them are being used as inputs to these models to ensure consistency between the steady-state and dynamic approaches. This has required some consultation and review of assumptions and methods used in the Steady-state Mass Balance model, for example in relation to estimating base cation release from rock phosphate fertiliser applied to plantation conifer forests.

The evaluation of the sodium dominance index was undertaken in collaboration with Dr Chris Evans and used data from MAGIC model runs (Evans *et al.*, 2001) funded previously by Defra under the 'Dynamic Modelling of Acidification Processes (EPG 1/3/133) and the 'Acidification of Freshwaters: The Role of Nitrogen and Prospects for Recovery' project (EPG 1/3/117).

5. Discussion and implications

This task has maintained a 'watching brief' on the application and development of steady-state critical loads in the UK and Europe. Development of the dynamic approach to critical loads modelling has also required review and evaluation of the data sets and assumptions underlying the steady-state approach in order to ensure consistency between the two approaches.

The evaluation of uncertainty has highlighted the necessity for greater transparency in the way uncertainties are derived so that communication between scientists and decision makers can be improved. Specific recommendations arising from the work are that parameter uncertainty should be defined exactly in terms of range, distribution and where appropriate scale and ecosystem type. There is a need for coordinated effort to agree upon uncertainty ranges for critical load input data so that a consistent approach is used across Europe. The work has been published in the journal 'Environmental Science and Policy' (Heywood *et al.,* 2006).

The NaDI method was proposed by White et al. (1999) as a relatively simple and widely applicable approach for estimating weathering rates from stream water chemistry. Evaluation of the method using a large regional data set has shown little overall advantage compared to using a regression model based on the sum of excess base cations in the stream water sample. In practical terms, the NaDI model is simpler to apply, only requiring the measurement of three base cations in a stream water sample whereas an additional measurement of chloride ion is required to calculate excess base cation concentrations. These results were presented as a poster (Reynolds *et al*., 2005) at the "Acid Rain 2005" conference in Prague.

6. Possible future work

It will be important in the future to retain links between the science developed in Terrestrial Umbrella projects and the UK NFC who have responsibilty for policy related applications such as future calls for data from the CCE and inputs to the revision of UK and UNECE emission control protocols. A small Task within the Terrestrial Umbrella is an effective way of ensuring communication between the scientists and modellers and the UK NFC.

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Work Package 1:
Refinement of Critical Loads

Task 2:
Refinement of empirical critical loads of nitrogen and their application to the UK

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Task 2 - Refinement of empirical critical loads of nitrogen and their application to the UK

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1. Summary

The aim of this Work Package was to develop a more objective approach to assessing empirical critical loads of nitrogen, using differentiated values within the range recommended for application within UNECE, based on ecosystem characteristics. The approach that has been developed, after active consultation with potential users, provides a decision support matrix (DSM) based on a combination of data from the National Vegetation Classification (NVC), site-specific information where appropriate, and endorsement theory to assess the strength of evidence supporting a particular recommended critical load value. The method has received broad support from major potential users, including Defra, the Joint Nature Conservation Committee (JNCC) and the Scottish Environment Protection Agency (SEPA). The decision support matrix has been applied to generate modified empirical critical loads for UK heathlands. Overall, the DSM supported the use of critical loads towards the lower end of the range recommended by UNECE, which is consistent with the current UK mapping value for this habitat. Since many NVC communities are located in specific regions of the UK, the DSM can be considered by Defra for wider qualitative application in national risk assessments.

2. Policy Relevance

Definition of empirical critical loads of nitrogen deposition as a range of values, with associated levels of certainty, is an important conceptual development in risk assessment based on critical-loads. However, application of this information in a policy context is poorly developed. For instance, in national critical loads risk assessment for the UK, a single critical load mapping value has been selected within the range recommended by the Berne workshop (Bobbink et al., 2002), based on expert judgment, and used to assess critical load exceedance. Because deposition rates often lie within the empirical critical load range, decisions on whether a site or grid square has a critical load exceedance may be highly sensitive to the choice of mapping value. This work package, although limited in scope, has produced a framework DSM that provides a more objective and transparent basis for decisions on the most appropriate critical load within the range of values recommended for a particular habitat. Since it is based on NVC communities, the approach can be used both for national and regional scale risk assessment, and for site-specific evaluation.

3. Objectives

The objectives for this task were:-

- To develop a Decision Support Matrix to facilitate selection of site-relevant empirical critical loads of nitrogen deposition
- To assess if any modifications are required in the mapping values of empirical critical loads of nitrogen deposition currently applied in the UK

4. Methods and results

A framework for the DSM was initially developed based on a consultation workshop with potential users, including Defra, JNCC, the Environment Agency (EA) and SEPA, in 2005. At the consultation meeting, significant problems were identified by agency users in interpreting the modifying factors for empirical critical loads identified at the Berne workshop, because of their limited knowledge and because of the lack of site-specific data on edaphic and climatic conditions. We therefore adopted an approach, which emerged in discussion at the workshop, which links the DSM strongly to NVC categories, since this information would readily be available to agency users. The use of the NVC has additional benefits since:- it is a standardized database for the whole country, that can be automatically linked to the EUNIS categories used to define critical loads through the NBN national habitats directory; NVC categories are normally identified in site notifications; NVC categories reflect major climatic and edaphic gradients; and NVC databases can be readily be linked to information on component species, e.g. through the PLANTATT database. However, for site-specific application, this assessment based on NVC categories needs to be complemented by information provided by the user.

Based on valuable input from Richard Wadsworth and Jane Hall (CEH Monks Wood), endorsement theory was used to combine the information provided for NVC categories and, when appropriate, by the user to identify the most appropriate critical load range from three options (lower end, mid-range, and upper end of 'Berne' values), with an indication of the strength of the underlying evidence (Wadsworth & Hall, in press). The workshop considered that using the critical load range in this way was more meaningful than using a single value given the uncertainties in data. This is a very important conclusion from the workshop, which has led to considerable modification of our planned approach and has wider implications for risk assessment using critical loads.

Table 2.1 summarises the information which is entered into the DSM as a basis for the endorsement procedure to determine which part of the critical load range should be used. The PLANTATT database is used to calculate averages for the Ellenberg nutrient, acidity and moisture indices for each NVC category. It is assumed that NVC categories with a low nutrient index, a high acidity index and a higher proportion of lichens and bryophytes would have a lower critical load, and that sites with management leading to lower N removal would also have a lower critical load. The habitat status and presence of rare species are optional inputs which allow the user to take a more precautionary approach.

Table 2.1 Summary of Variables used to define Critical Load range

Variable	How defined	Link to user
Nitrogen status	Linked to Ellenberg nutrient index by NVC community	User defines NVC category
Acidity status	Linked to Ellenberg acidity index by NVC community	User defines NVC category
Moisture status	Linked to Ellenberg moisture index by NVC community	User defines NVC category
Proportion of lichens/bryophytes	Determined from species data for NVC community	User defines NVC category
Presence of rare species	Based on site-specific information	User defines number of rare plant species
Management of site	Based on site-specific information and history	User identifies from habitat-specific options
Habitat status	Based on Common Standards Monitoring for site	User defines site condition as 'favourable' or 'unfavourable' etc

As the species listed for each NVC community, and linked to the PLANTATT database, vary in frequency from 'constants' to 'very occasional', only higher plants found in a community with a 'frequency/constancy' of greater than II (40%) were included in the calculations. 'Frequency' refers to how often a plant is found on moving from one sample of the vegetation to the next, irrespective of the abundance of that species in each sample.

Table 2.2 shows how the 'strength' or 'weight' of evidence is allocated for the endorsement procedure.

Table 2.2 'Strength' of evidence used in the endorsement procedure

Variable	How defined	'Strength' of Evidence
1. Nitrogen status	Ellenberg N average calculated for 'n' higher plant species in a community with a frequency of occurrence > 40%	n = < 5 – 'weak' n = 5-10 – 'medium' n = >10 – 'strong'
2. Base cation status	Ellenberg R average calculated for 'n' higher plant species in a community with a frequency of occurrence > 40%	n = < 5 – 'weak' n = 5-10 – 'medium' n = >10 – 'strong'
3. Moisture status	Ellenberg F average calculated for 'n' higher plant species in a community with a frequency of occurrence > 40%	n = < 5 – 'weak' n = 5-10 – 'medium' n = >10 – 'strong'
4. Proportion of lichens/bryophytes	Assumed that lower plants are more susceptible to N impacts	'medium'
5. Presence of rare species	User defined from pull down list	'strong' for rare higher plants that have Ellenberg values in PLANTATT; 'medium' for lower plants.
6. Management of site	User defined from pull down list	'strong'
7. Habitat status	User defined from pull down list	'strong'

As the aim was to develop an approach that can potentially contribute to national critical load mapping, as well as at a local site-specific level, every effort was made to make the DSM approach to determining which part of the critical load range to apply as compatible as possible with the modifying factors considered by the Berne review (Bobbink et al. 2002). Reasons for omitting temperature and P limitation, identified as important by Bobbink et al. (2002) as modifying factors, in the DSM are described below.

Despite the fact that the consultation meeting decided that including temperature as a modifier was a possibility for UK sites, in the DSM it is considered to be a secondary factor for the modification of critical loads as NVC communities will correspond with specific climate zones within the UK; furthermore, low temperatures will reduce N availability and this should be reflected in the nutrient status based on plant species composition. In addition, temperature and water status of sites can also result in contradictory modifications of the critical load e.g. upland sites in the UK can be both 'wet' and 'cold'. The consultation meeting also considered that the inclusion of frost hollows and extremes of temperature was problematic.

The consultation meeting also concluded that generally only very qualitative data are available on P availability. It was suggested that if specific data on soil P levels or N/P ratios in vegetation are not available, other indicators may be used, but these are not usually available at sites. Therefore, no recommendation can be made at present for inclusion of a specific modification related to P limitation. Furthermore, there is recent UK experimental evidence for both heathlands and grasslands, summarised in this contract report, to suggest that increased N deposition can affect P-limited systems, contrary to the assumptions

underlying the Berne recommendations. In the application of the DSM to heathlands, the Ellenberg indicator for soil moisture (F) was originally included, but the results indicated that the range of conditions under which species are usually found in the UK give a low range of F values which do not assist in discriminating critical loads.

Having developed this conceptual framework for the DSM, it was then applied to UK heathlands. A document summarising the approach and its application to heathlands was circulated to the user group of workshop participants with a questionnaire requesting comments and feedback.

The results (Table 2.3) of applying this scheme to the four variables obtained by interrogating the NVC and PLANTATT databases show that there is no 'definitive' or 'confident' case for changing the critical load for any heathland classification from the mid–point. However, there are 2 cases out of 22 where it is 'likely' that the higher part of the critical load range should be applied: These are:- H5 (*Erica vagans - Schoenus nigricans* heath; transitional between the lowland dry heaths and the wetter communities classified in the NVC as mires) and H6 (*Erica vagans - Ulex europaeus* heath; lowland dry heath).

The only 'definitive' case for keeping the mid-point of the range was for H18 (*Vaccinium myrtillus – Deschampsia flexuosa* heath), and there were two 'likely' cases for the mid point: H6 (*Erica vagans - Ulex europaeus* heath; lowland dry heath) and H20 (*Vaccinium myrtillus-Racomitrium lanuginosum* heath).

There are 15 cases of the 22 NVC communities for which it is 'possible' that the lower part of the critical load range should be applied, in four of these it is equally 'possible' that the midpoint should be used. On balance, therefore, the application of the DSM leads to the conclusion that it is possible that the critical load for this habitat should be set towards the lower part of the range of empirical critical loads recommended (10-20 kg N ha^{-1} yr^{-1}).

These results can be supplemented by information on the other three user defined variables: presence of rare plants, management applied at site and the habitat status from Common Standards Monitoring at the site (see Table 2.1). This additional information will be particularly usefully in cases where the use of the four NVC based indices produces an equal likelihood of two options.

Table 2.3. Summary of Results of Endorsement Procedure applied to 22 NVC Heathland Communities

	'Definitive'	'Confident'	'Likely'	'Possible'	'Weak'
Use higher part of CL range:			2		
Use mid point of CL range	1		2	2	
Use lower part of CL range				15	

Note: "Definitive", "three or more pieces of ""strong"" evidence and no conflicting evidence"; "Confident", "two or more piece of ""strong"" evidence no ""strong"" evidence for any alternative"; "Likely", "at least one piece of ""strong"" evidence, other alternatives may have ""strong"" evidence"; "Possible", "at least one ""moderate"" or two ""weak"" pieces of evidence (other alternatives may have greater weight)";"Weak", "some evidence."

5. Discussion and implications

There are two important aspects of the outcome of this project. The first is the degree to which the approach in our DSM could be adopted and applied by agencies, including Defra. The consensus view from the DSM questionnaire responses was that the proposed approach would be beneficial for the regulatory decision on potential impacts, helping to both narrow down some of the uncertainty associated with the critical load approach, while also placing the UK mapping data into a more site-specific context. The JNCC, whose response incorporated comments form Natural England, Countryside Council for Wales and the Environment and Heritage Service (Northern Ireland) stated that there are a number of key aspects that could benefit, including casework advice, supporting/interpreting site condition reporting, management planning and Favourable Conservation Status (FCS) for the Habitats Directive. SEPA added that the DSM could benefit those that deal with regulated site-specific case work to identify potential critical load exceedance and place a scoring potential for the degree of risk to specific habitat sites. JNCC and SEPA strongly encouraged the application of the DSM to case studies and gave advice on how to improve the DSM approach, especially regarding the inclusion of management issues, habitat status, rare species, lower plants, site history and the interpretation of results from the endorsement procedure.

The Environment Agency favoured an approach to site specific critical loads for nutrient nitrogen based on specifying each term in a mass balance equation of an ecosystem (e.g. as described in Skeffington et al. (2007)). They acknowledge the difficulty in specifying the values of the terms in the mass balance equation, but view the usefulness of the NVC approach as a way of supplementing available information and the endorsement theory as a way of using surrogate information based on expert judgement in a systematic way. Application of the mass balance method for nutrient nitrogen at two test sites revealed that the method was most sensitive to the definition of acceptable nitrogen leaching and that for exceeedance, deposition parameters were more important at high deposition/low critical load sites.

Our view is that the DSM can provide a transparent and objective way of applying empirical critical loads in the UK, which is more user friendly, and therefore of more general use, than the mass balance approach with its more stringent data requirements and associated uncertainties. The fact that deposition rates often lie within the empirical critical load range make it even more important that users at specific sites can make informed decisions based on information that is easily understood.

The second aspect, which is of more direct relevance to Defra, is the application of the DSM approach to support national mapping exercises. The results from the heathland exercise allow some comment on this point. Firstly, it is important to note that, overall, the DSM results suggest that the UK mapping value should be towards the lower end of the empirical critical load range for this habitat. In fact, this is consistent with current practice, since the UK applies, based on a specific experimental study, a critical load of 12 kg N ha^{-1} yr^{-1}, rather than the value of 15 kg ha^{-1} yr^{-1}, which is the mid-point of the recommended range.

The geographical distribution of the NVC communities which the DSM suggests should have a higher or lower empirical critical load is also important. The two NVC communities for which a critical load towards the higher end of the range was ‘likely’, H5 and H6, are both restricted to Cornwall and Devon, and hence this would not have a major impact on national exceedance maps. In contrast, NVC communities for which the DSM suggests a critical load

towards the lower end of the range is possible include those found in East Anglia, Wales, the Midlands, and Scotland. This wide distribution again supports the national application of an empirical end towards the lower end of the range.

6. Possible future work

This DSM approach which has been developed under this Work Package has the potential to be developed and applied in a range of contexts. For Defra, the most important application is likely to be an improved method for national mapping of empirical critical loads of nutrient nitrogen. At present, expert judgment is used to decide what value, within the UNECE range of empirical critical load values for each habitat, should be used for national mapping. The DSM provides a more objective basis for these judgments and, by linking the assessment to NVC data, offers the potential for developing mapping values of empirical critical loads which vary in different regions of the UK. A further option which has been discussed as part of the development of the DSM is to link the likelihood function of the habitat- or site-specific critical load to the probability function for the modeled rate of nitrogen deposition. This would allow an estimate to be made of the likelihood of critical load exceedance, either at a specific site, or as part of a national mapping exercise.

In terms of other agencies, future applications of the DSM could include assessment of the risk of impacts of nitrogen deposition on individual sites under the Habitats Directive, and evaluation of the condition and future threats to the integrity of SSSIs. To facilitate wider application by a range of users, it would be possible to develop an electronic version of the DSM which would generate the likelihood of a given part of the empirical load for a particular site from appropriate data. Such a development would also allow addition of other variables, and refinement of the approach (e.g. by assigning nutrient and acidity values to lichens and bryophytes (not available in PLANTATT), and to rare species). Finally, Richard Wadsworth and Jane Hall (CEH Monks Wood) have been developing an endorsement based approach for critical loads of nutrient nitrogen using a mass balance approach. There would be considerable value in comparing, and possibly combining, the two approaches to provide a stronger basis for assessing appropriate site-specific critical loads.

This approach will not be supported under the new 'Terrestrial Umbrella' programme, and we will seek to support further development of the DSM from other sources.

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Work Package 2:
Impacts, Recovery and Processes

Task 3:
Grassland soil and vegetation responses following nitrogen saturation at Wardlow Hay-Cop

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Task 3 - Grassland soil and vegetation responses following nitrogen saturation at Wardlow Hay-Cop

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1. Summary

Wardlow Hay-Cop is located in the Derbyshire Dales National Nature Reserve, a highly N polluted region that has received a total of over 2000 Kg N ha^{-1} ambient deposition from 1900-2000 (Fowler *et al.,* 2004), a value matched or exceeded on less than 10% of the total area of the UK. Field plots established on the site in 1990 and 1995 have been used to investigate the long term effects of enhanced atmospheric N deposition (0-140 kg N ha^{-1} y^{-1}) on soil and vegetation in adjacent species-rich semi-natural acidic and limestone grasslands. Since 2002 the earlier plots are in 'recovery', and since 2004 the later set of plots are now divided, one half continuing to receive treatment, the other being in recovery.

The studies carried out in this contract included measurements of (1) seasonal and annual changes in extractable N pools in plots treated for 12 years, that have been in 'recovery' since 2002, (2) changes in vegetation and soil properties in plots established in 1995 which continue to receive N treatments, including effects of N treatment interaction with phosphorus supply, (3) the effects of ceasing, from 2004, N inputs to half the area of these plots, to determine the extent of recovery of soil and vegetation variables, and the legacy effects of the N and P treatments.

Key findings from the present work include: (1) Soil extractable N concentrations have almost completely recovered within 4 years in the limestone grassland, whereas in the acid grassland eutrophication is more serious, leaving a more persistent legacy of increased ammonium concentrations despite ongoing grazing removal of N from plant biomass. Extractable mineral N concentrations in the acid grassland remain up to twice those in control, untreated plots four years after ceasing N treatment. (2) Soil acidification has occurred in response to 140 kg N ha^{-1} y^{-1}. After 10 years, soil pH decreased from 6.3 to 5.4 in the limestone grassland, and from pH 4.5 to 4.2 in the acid grassland. In the two grasslands up to 46% and 18% of the available calcium, and 33% and 63% of the available potassium have been lost from surface soil horizons, and some plant species show marked decreases in concentration of these elements in their foliage, with little evidence of recovery. (3) The flowering of forbs is seriously reduced by excessive N deposition, and shows no significant recovery in three growing seasons after ceasing experimental treatments. The interactions between N and P availability are species-specific, some showing no effects of increased P availability, whereas other species show positive or negative responses.

2. Policy Relevance

Grasslands are the most extensive semi-natural plant communities in the UK countryside, and over 20% of our native higher plants are specifically associated with these habitats (Preston *et al.,* 2002). Acidic and calcareous grasslands cover 1.23 million hectares of the UK and are of

exceptional conservation and amenity value as a result of their floristic diversity, containing over 300 of the 540 native plants specifically associated with grassland (Preston *et al.,* 2002). Large areas of our semi-natural grasslands have experienced several decades of N deposition in excess of their currently defined critical loads- 10-20 kg ha y^{-1} (Achermann & Bobbink, 2003). and there is increasing evidence that these grasslands have suffered major loss of diversity and habitat quality as a result (Smart *et al.,* 2004, Stevens *et al.,* 2006). Two key policy issues arise from these findings:

(a) The need for robust evidence of either continuing environmental damage or of recovery being constrained by current N deposition loads, to justify establishment of further cuts in emission targets- especially for ammonia- which remains at historically high values.
(b) The need to establish whether recovery from the habitat damage due to N enrichment is constrained by long-lasting impacts that may require active intervention (e.g. neutralization of soil acidity and replenishment of base cations; seeding of species that have declined).

We have found that N pollution on grassland causes legacy effects that include eutrophication, soil acidification and base depletion, biodiversity loss, reduced flowering of forbs, and increases in abundance of common sedges and grasses at the expense of forbs and bryophytes. The flowering of forbs is especially sensitive to N deposition- seriously compromising the quality of the habitats through impoverishment of the seed-bank, gene pool, and potentially reduced resilience to stresses such as climate change.

An urgent priority is to establish which of the nitrogen pollution impacts leave the most persistent legacies and what remedial actions may be required to restore N-damaged grasslands. This information is particularly required by Natural England, who manage many species-rich grassland nature reserves, and by DEFRA both to inform policies for emissions target setting and for assisting farmers and landowners in better managing grasslands damaged by N deposition. The plots at Wardlow Hay-Cop provide the first data to enable analysis of the long-term impacts of N deposition on limestone and acidic grasslands, and their recovery from these impacts- providing the crucial data required inform future emissions and management policies affecting these two major types of grassland.

3. Objectives

The current phase of the long-term studies of grassland soil and vegetation responses to nitrogen saturation based at the Wardlow Hay-Cop site had the following four objectives

1. Evaluation of the potential chemical and biological impacts of nitrogen deposition.
2. Evaluation of the recovery of damaged ecosystem and interaction effects with P status.
3. Assessment of the processes controlling nitrate leaching and the rate of leaching.
4. Collaboration with researchers under the Freshwater and Dynamic modeling Umbrellas.

4. Methods and results

Wardlow Hay-Cop in the Derbyshire Dales National Nature reserve, is in region that for the past 50 years has experienced some of the highest rates of N and sulphur pollutant deposition in the UK. It still exceeds the critical N loads defined for such ecosystems.

Plots were first established on acidic and limestone grassland in 1990 and received approximately monthly additions of NH_4NO_3 solutions as a fine spray to enhance the nitrogen deposition by 0, 35, 70 and 140 Kg ha^{-1} y^{-1}. In July 2002 the treatments on these plots were stopped to allow assessment of recovery processes. In 1995 a further series of larger ($9m^2$) plots was established on these grasslands and are treated with 0, 35 and 140 kg N ha^{-1} y^{-1} as NH_4NO_3 with and without phosphorus (as NaH_2PO_4) at 35 Kg P ha^{-1} y^{-1}. The P additions to the limestone grassland was stopped after 1 year due to major floristic changes, but all other treatments, including N additions to these formerly P treated plots, have continued. In August 2004 the plots established in 1995 were subdivided, one half of each continuing to receive treatments, the other being used to study recovery.

4.1 The legacy of N enrichment- effects on extractable mineral N pools in soil

After 12 years of treatment, the plots established in 1990 have been in 'recovery' since August 2002. We have continued to monitor the concentration of ammonium and oxidized N (nitrate) extracted by KCl from the soil through the period February-June each year (Fig. 1) as the peak annual concentrations of both ammonium and nitrate occur during this period. Almost complete immobilization of mineral N occurred even in the highest N treatments in August. After ceasing treatments the mineral N pools in both soils have rapidly declined with no detectable enrichment remaining in total extractable mineral N in the limestone grassland. There are residual effects of the previous N loads on extractable mineral N in the acid grassland (particularly of ammonium), but these are modest by comparison to the mineral N pools when the plots were being treated (Fig. 1). Overall, the prospects for recovery of soil mineral-N pools following reductions in N deposition are encouraging, but the rate of recovery is declining (Fig. 1).

In both grasslands total oxidized nitrogen concentration (primarily nitrate) remains double in plots that were treated with 140 kg N ha^{-1} y^{-1} than in control plots, 4 years into recovery, but these effects are not significant ($P>0.05$). Nitrate concentrations are up to ten-fold higher in the limestone grassland than the acid grassland, and nitrate leaching may contribute significantly to the export of N from the former, assisting 'recovery'.

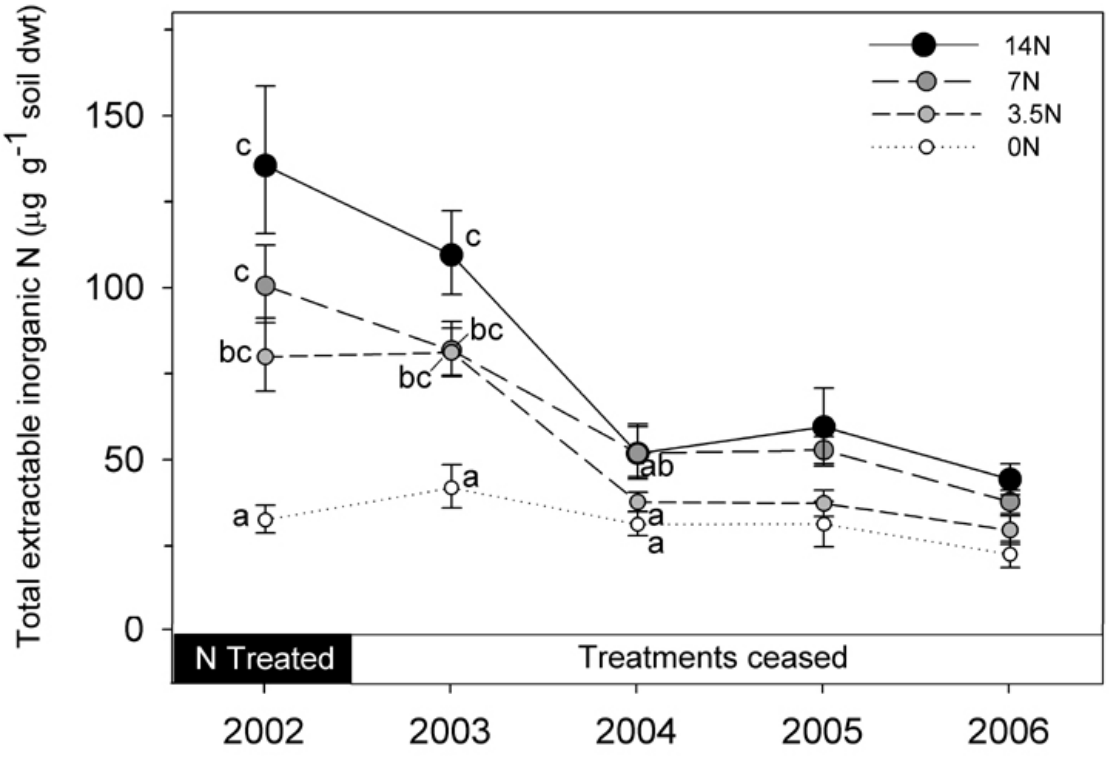

Figure 1. Mean total extractable inorganic N concentration in acid grassland soil after 12 years of monthly N treatment applications (2002) and in 4 years after ceasing treatments (0N = control, 3.5N = 35 Kg N ha^{-1} y^{-1}, 7N = 70 Kg N ha^{-1} y^{-1}, 14N = 140 Kg N ha^{-1} y^{-1}. Error bars = 1 SEM, n=3 plots. Data is derived from bi-monthly samples from Feb-June each year.

In the acid grassland in the fourth year of recovery, soil extractable N concentrations remain up to twice those in control, untreated plots, and there remains a clear dose-related response to the former N treatments compared to that in the control plots that have never received our N treatments (Fig. 2).

Although some off-take of N from the enriched plots would therefore be expected through grazing, most of the added N is retained in the soil, and is not readily extractable. Most of the retained N must be in non-exchangeable mineral N and organic N, the latter likely to be the main sink as the organic matter content of the limestone and acid grassland soils are 32% and 40% of the dry weight respectively.

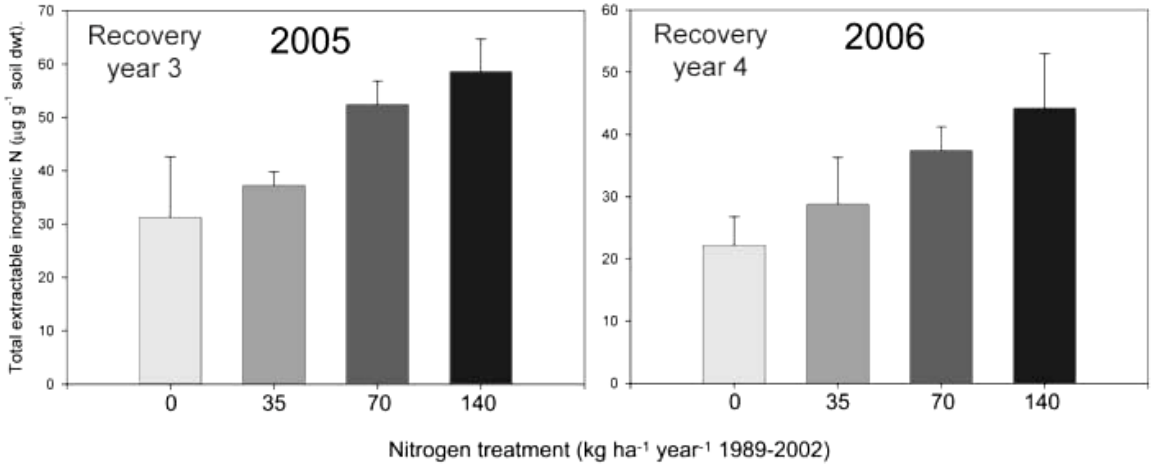

Figure 2. Mean total extractable inorganic N concentration in acid grassland soil, after 3 and 4 years 'recovery' following 12 years of monthly N treatment applications. Error bars = 1 SEM, n=3 plots. Data are annual means of bi-monthly samples taken Feb-June each year. Former nitrogen treatments are significant in both years (ANOVA $P<0.05$).

Our analysis of exchangeable soil N pools provides encouraging evidence that semi-natural grasslands that have experienced extreme N saturation can quickly recover towards the concentrations seen in soil receiving much lower N inputs. The rapidity and extent of recovery we have seen indicates that soil N availability is unlikely to present a major long-term constraint on floristic and overall habitat recovery. We must add two notes of caution. Our experimental plots are a small area in a large grazed field which is likely to result in progressive export of N from our plots, whereas in N enriched grasslands at landscape scale a greater proportional return of N will occur in dung and urine. Persistent N enrichment is occurring in the acid grassland, the effect is dominated by ammonium which remains the major form of pollutant N deposition, and floristic richness of acid grasslands have been shown to be most sensitive to eutrophication by N (Stevens *et al.,* 2006).

4.2 N deposition causes soil acidification

After 10 years of N application at 140 Kg ha^{-1} y^{-1}, there has been significant acidification of the soil in the limestone grassland and in both the organic and mineral horizons of the acid grassland (Fig.3). These data provide additional evidence that acidification of the limestone grassland has happened even in untreated plots due to sustained ambient acid deposition. The pH of the limestone grassland at Wardlow was 7.5 in the early 1950's (Balme, 1953). Measurements made in 1998, 8 years after establishing the original plots, and after a period from the 1950's-1980's in which the area experienced some of the highest UK rates of acid deposition from oxides of sulphur and nitrogen, showed that the pH of untreated plots had fallen to pH 6.8. By 2005 this has now fallen to pH 6.6, suggesting that soil acidification is continuing in these upland grasslands. The treated plots have seen more rapid acidification, confirming the contributory role of N deposition. After 8 years of the highest treatment to the limestone grassland plots established in 1990, the pH decreased to 6.1 (Johnson 1998), falling to pH 5.8 by the 10^{th} year of the plots established in 1995 (Fig. 3).

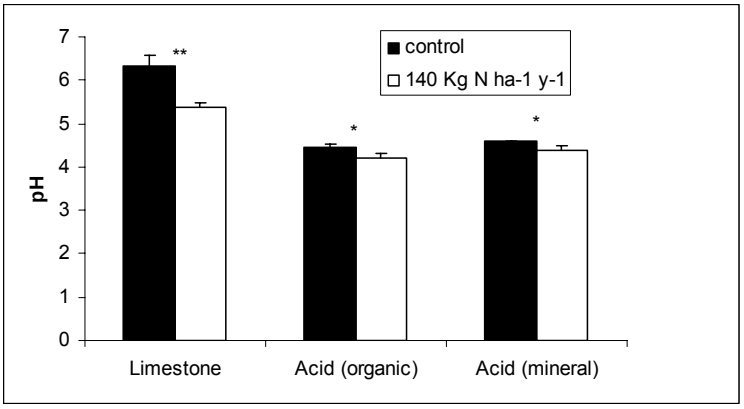

Figure 3. Mean pH measured in distilled water for soil sampled in April 2005 (treatment year 10) from control and treatment plots receiving 140 Kg N ha^{-1} y^{-1} in limestone and acidic grassland (organic and mineral horizons) at Wardlow Hay-Cop. N=3 plots, bars indicate SEM, * = $P<0.05$, ** = $P<0.01$ (ANOVA).

4.3 N deposition depletes base cations

Most of the N deposited on upland grassland is retained in the soil, even when the amounts deposited exceed plant uptake, and this soil reservoir is dominated by ammonium (NH_4^+) which displaces and exchanges with other cations leading to substantial loss of base cations. We have characterised the extent of base-cation depletion specifically from plant-available pools in the soil, carrying out extractions using water, ammonium nitrate and organic acids of the kinds secreted by roots of limestone and acidic grassland species (Tyler & Strom 1995) and measuring base (Ca, Mg, Na, K) and non-base (Fe and Al), cation concentrations (Table 1). Plant roots secrete low molecular weight carboxylic acids as a response to phosphorus and iron limitation, in Ca rich soils, and as a mechanism to protect against aluminium and iron toxicity in acid mineral soils (Tyler & Ström, 1995). Citric acid is particularly important for limestone grassland plants, whereas oxalic acid is more important for acid grassland species (Tyler & Ström, 1995).

In the limestone grassland there has been loss of between 12-46% of the base cations in aqueous solution and in the easily exchangeable pools. The most important being the loss of Ca which is the dominant cation in this soil. Although not statistically significant, the loss of 27-33% of the most available K pools in the limestone grassland is also of considerable concern given its importance as a plant nutrient. In the same grassland the non-base cation concentrations have generally increased by 0-34% depending on the ion and extractant, and these increases in Fe and Al are indicative of the acidification of this soil with non-base cations gaining increasing solubility.

Table 1. Percentage change in extractable cations in response to 11 years of treatment with 140 Kg N ha^{-1} y^{-1}. Extractants include water, 1 M NH_4NO_3 and 2mM citric and oxalic acids. Significant differences are indicated in bold typeface (ANOVA $P<0.05$).

	Limestone grassland			Acid grassland mineral horizon		
	Extractant			Extractant		
Element	H_2O	NH_4NO_3	Citric acid	H_2O	NH_4NO_3	Oxalic acid
Ca	**-46%**	**-22%**	**-27%**	-18%	+19%	+11%
Mg	-19%	-28%	**-36%**	-37%	-11%	-23%
Na	-44%	-26%	-38%	**+33%**	+6%	-40%
K	-33%	-27%	-12%	-60%	-31%	**-63%**
Fe	+2%	+9%	**+33%**	+24%	-6%	-14%
Al	0%	+34%	+18%	-6%	-6%	+10%

In the acid grassland the concentration of bases is much lower and the effects of the N treatment less consistent than in the limestone grassland. Surprisingly, and in marked contrast to the limestone grassland, there appears to be no consistent or major effect on Ca concentrations, but there is a consistent trend of Mg loss, but this is not significant. The most important cation loss from the acid grassland is that of easily extractable K which has declined by of 31-63% -presumably due to NH_4^+ exchanging for K^+ in the soil.

The soils in our most heavily N enriched plots have suffered both acidification and gross depletion of bases, particularly of K and Ca. These changes are likely to leave long legacy effects, and the reduced buffering capacity of the soils may render these ecosystems more sensitive and vulnerable to future N and acid deposition. Further studies are required to investigate the rates of recovery of these components of soil chemistry on reducing N inputs.

4.4 Effects of N enrichment on plant nutrient status and recovery

Foliar analysis of selected species in both grasslands revealed that K concentrations were reduced by over 25% in *Sanguisorba minor* and *Potentilla erecta* in the 140 Kg N ha^{-1} y^{-1} plots in the limestone and acid grasslands respectively. However, co-occurring species (*Thymus polytrichus*, *Carex flacca* and *Helianthemum nummularium*) showed no effects of the N treatment on K concentrations. Analysis of mixed species 'turf' samples from the limestone grassland detected significant decrease in Ca concentration in foliage by 47%, however, the reduction in Ca concentration in selected species was much less: *Sanguisorba* –19%, *Thymus* –17% and *Carex* –12%. The greater decrease in the whole turf reflects a shift to greater dominance by grass and sedges that have lower Ca concentrations than most of the forbs. For the majority of species we have examined there is little evidence that recovery in foliar cation concentrations is occurring in response to ceasing N treatments. Overall, our results demonstrate that N enrichment can have important impacts on foliar chemistry, and that the loss of exchangeable soil cations is being reflected in foliage of some, but not all, species.

4.5 Effects of N enrichment on plants- the role of soil P status and recovery

The recovery sub-plots established from division of the 1995 plots in 2004 have allowed pair-wise comparisons to be made between adjacent parts of the same plots: one half receiving N additions and the other half in recovery. This experimental design is very sensitive to detecting small changes in floristic composition and permits normalization to account for inter-annual variation. In the present contract we have focused attention on obtaining data for N treatment effect on selected, important species. The point-quadrat methods used previously had proved too insensitive to detect significant effects on cover of most species, so instead, the numbers of shoots per unit area or cover has been recorded for selected species (*Carex flacca, Conopodium majus* and *Anemone nemorosa*). Whilst *Conopodium majus* shoot density has decreased by 88% in the highest N treatment, there is no evidence of any recovery 4 years after ceasing treatments. The cover of Anemone nemorosa has decreased by over 80% under the highest N loading, but does show some early evidence of recovery in the 4th year after ceasing treatment, but this is not significant (P>0.05).

Carex flacca is a dominant component of the limestone grassland and has proved highly resilient to N enrichment, and has tended to increase in dominance as other species have declined (Figure 4). However, *Carex flacca* is very adversely affected by increased phosphorus availability- treatment with 35 Kg P ha^{-1} for the first year of the plots established in 1995 almost eliminated the species due to competition with grass, and in the following 12 years in which no further P has been added to the plots it is clear that enhanced N deposition has enabled recovery of the sedge in a clearly dose-dependant manner (Figure 5). This recovery has been slowed in the sub-plots that from 2004 no longer receive supplemental N.

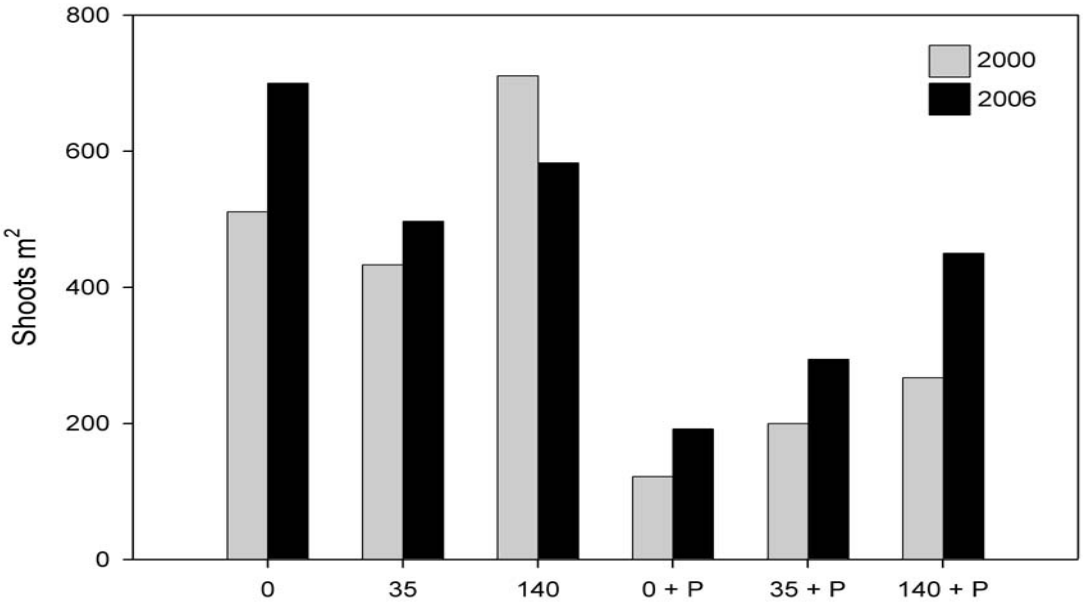

Figure 4. Mean number of *Carex flacca* shoots per m^2 in relation to N and P treatments of field plots in the years 2000 and 2006. (0 = control, 35 = 35 Kg N ha^{-1} y^{-1}, 140 = 140 Kg N ha^{-1} y^{-1}; +P = 35 Kg ha^{-1} P added in the year 1995-1996.

One of the most important impacts of N deposition is reduced flowering of forb species in both acid and calcareous grassland. Many forbs show proportionally greater reductions in flowering than in cover in response to high N deposition- so that flowering is a particularly sensitive indicator of N impacts. The flowering of forbs is a key component of the amenity

value of semi-natural grasslands. Reduced flowering seriously diminishes the biological and aesthetic value of species-rich grassland. Due to the particular sensitivity of flower production to N deposition we have carried out detailed studies with the aim of being able to detect any early evidence of recovery on ceasing experimental N additions to the grasslands. Important forb components of both grasslands show clear does-dependent declines in flowering in response to N inputs (Fig. 5). To date, we have not detected any significant improvement in flowering in the recovery sub-plots. However, there are encouraging indications that some species including *Scabiosa columbaria* in the limestone grassland and *Potentilla erecta* in the acid grassland are starting to recover 1-3 years after ceasing N treatments. Longer-term monitoring will be required to establish if these trends are real and are sustained.

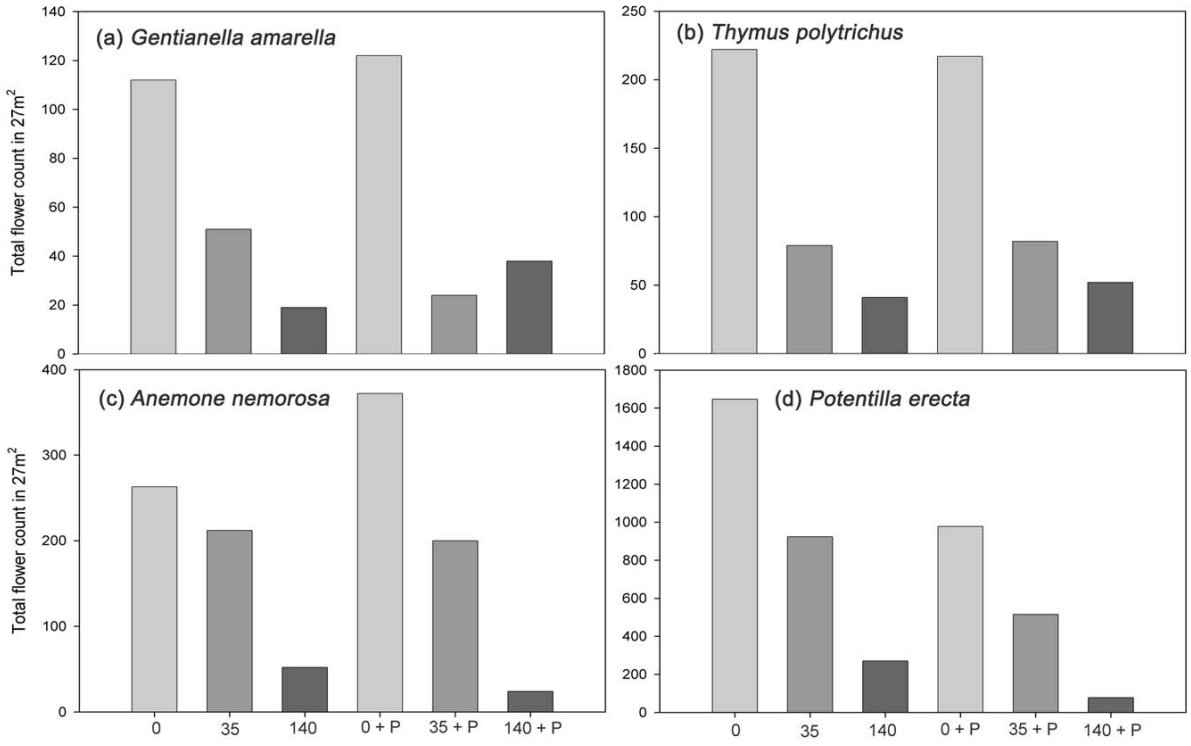

Figure 5. Total flower counts for selected species in relation to N and P treatments of field plots in the limestone grassland in 2005 (a), (b); and in the acid grassland in 2006 (c), (d). (0 = control, 35 = 35 Kg N ha^{-1} y^{-1}, 140 = 140 Kg N ha^{-1} y^{-1}; +P = 35 Kg ha^{-1} P only added in the year 1995-1996 in the limestone grassland plots.

Increased phosphorus availability protects some species (e.g. *Conopodium majus*) against impacts of N deposition, whereas in other species it has no apparent effect (e.g.*Thymus polytrichus*- Fig. 5 b) while in a third group it compounds the impacts of N (e.g. *Potentilla erecta*- Fig. 5 c). The interactions between N and P availability are therefore complex and species-specific.

5. Discussion and implications

Setting critical loads for species-rich grasslands has proved difficult as floristic changes due to nitrogen enrichment treatments have occurred slowly, and even when the magnitude of effects are large, they are often not statistically significant because of the high inherent spatial and inter-annual variability in these communities. It is only in very long-term studies, such as ours, that effects can be detected. Because change is slow it might be falsely assumed to be unimportant. However, once damage has occurred, recovery is also slow so avoiding or minimizing damage in the first place is crucial. Despite major progress in national emissions controls many upland grasslands continue to receive N deposition rates that are above the critical limits for these ecosystems, and further cuts in emissions are required to protect these ecosystems from further damage. The standard measures used to monitor plant communities such as cover estimates and presence or absence of species are much less sensitive to N deposition than is the production of flowers, or direct measures of plant abundance such as numbers of shoots per m^2 and we have adopted these more sensitive sampling methods. As a result, the full extent of N deposition impacts on plant communities will have been underestimated by national vegetation surveys and monitoring. Much of the change in grassland communities that are N saturated is cryptic- and not directly visible. This includes changes in soil chemistry such as acidification, loss of base cations, eutrophication and enrichment in available ammonium and nitrate pools. The extent of soil chemical changes are very large, and serious acidification and base depletion we have detected appear to leave major legacy effects that may impact the ecosystems for decades. Persistent eutrophication is a particular problem of acidic grassland due to the accumulation of a large pool of exchangeable ammonium in the soil. Further work is required to determine whether this is a widespread problem in upland grasslands or is mainly associated with a sub-set of soil types.

Our results are important for understanding the impacts of N on grassland ecosystems and their functioning and the timescales over which any recovery may occur in response to increasingly stringent emissions controls. The data we have obtained and submitted to the coordinated database are of importance both for the Dynamic Modeling Umbrella and Freshwater Umbrella projects in helping to understand and model the fate of N deposition in terrestrial and aquatic ecosystems.

Major findings of this work package have been presented at National meetings (CAPER 2006, 2007), and at annual meeting of the Terrestrial Umbrella Programme (2005, 2006), and a PhD thesis based on the results is in preparation (Mr O O'Sullivan).

6. Possible future work

6.1 Indicators of N deposition impacts and for monitoring recovery

In the next phase of the project we propose to conduct studies to establish the most ecologically-relevant indicators of sensitivity to N deposition and of ecosystem change due to N saturation. We propose to study responses of potential indicator species in the treated plots in terms of abundance, foliar nutrient status, N:P ratios and enzymes associated with nutrient uptake. Soil parameters will include extractable N, N:P ratios, enzymes associated with nutrient uptake and other chemical changes including base cation:ammonium index, base saturation, and pH. The suitability of the best of these indicators in the experimental plots for application to national monitoring will be investigated in studies of grasslands of the same

floristic type but in regions experiencing very different rates of N deposition. These studies will assist the development of appropriate indicators that can be used for national monitoring of N deposition impacts and any recovery from these impacts.

6.2 Data provision for modeling N impacts and recovery

We propose targeted data collection to provide the data required for modeling N impacts and recovery (Work Package 6 in the Terrestrial Umbrella contract 2007-2010). This work will combine synthesis of existing data obtained from the Wardlow grassland sites, together with collection of additional data required for the modeling.

6.3 Chronology of recovery of floristic and soil parameters

We propose to continue to monitor floristic changes and post-treatment recovery in selected plant species (flowering and shoot density per m^2), and to maintain long-term monitoring of soil chemical changes driven by enhanced N inputs, and in untreated subplots determine the rates by which these parameters show recovery (pH, base cation status, extractable mineral nitrogen).

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Outputs from this project

O'Sullivan O. (to be submitted 2007-8). Long-term impacts and recovery of grasslands from N deposition.

Work Package 2:
Impacts, Recovery and Processes

Task 4:
Long-term impacts of enhanced and reduced nitrogen deposition on semi-natural vegetation

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Manchester Metropolitan University

Task 4 - Long-term impacts of enhanced and reduced nitrogen deposition on semi-natural vegetation.

PIs: S.J.M. Caporn, J. Edmondson, J.A. Carroll, M. Pilkington, N. Ray

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1. Summary

Work under task 4 has progressed over the period 2004 – 2007, in line with the overall objective of evaluating the potential chemical and biological impacts of nitrogen deposition on moorland and heathland vegetation and soil systems.

Key findings include:

- Reductions in available N in litter and plant compartments only three years after halting N additions.
- Evidence of N saturation in lichens and bryophytes at N inputs at as little as 10 kg N ha^{-1} y^{-1} above ambient, leading to significant reductions in cover and diversity.
- Evidence of the value of: bryophyte diversity; litter extractable N; foliar N as bioindicators of N deposition and overall N status in wider regional surveys of *Calluna* moorland.
- Negative effects of long-term N loading on *Calluna* canopy regrowth following management (burning, cutting).
- Future work needs to (i) further integrate findings with soil-vegetation models and (ii) develop and validate potential bio-indicators for eutrophication at a national scale.

2. Policy Relevance

2.1 Early effects of reduced nitrogen loading

Previous work on the Ruabon moorland plots found large accumulations of N in peat and soil layers, suggesting that the accumulated soil N would be difficult to remove and recovery of the system could be slow. However, the recovery experiment on the new plots at Ruabon, shows that in more rapidly cycling soil/plant compartments, available N (nitrate and ammonium) may decline much more quickly in response to reduced deposition. It follows from this that beneficial effects could be seen on the vegetation structure of polluted moorlands in the first few years of reduced nitrogen loading.

2.2 Need for more frequent management

In earlier work we undertook controlled management by moorland burning and lowland heath cutting. Post-management re-growth of heather from the stem bases was slowed in the higher N treatments, following management, at both sites. This delayed re-growth after management is probably an indirect result of nitrogen-enhanced ageing of *Calluna* and could allow invasion and establishment of competitor plant species such as *Deschampsia flexuosa*.

This was found on the lowland heath experiment. More frequent management would therefore probably benefit heather-dominated communities on nitrogen affected upland and lowland heaths.

2.3 N/P limitation affects species differently

Critical loads for N with respect to bryophytes and lichens are potentially altered by availability of phosphorus. *Calluna* dominated moorlands are often N limited, and a shift to P limitation may indicate N saturation. Data from the new plots at Ruabon show a shift to P limitation in the growth of the lichens and bryophytes at very low levels of nitrogen input, at which *Calluna* growth is still strongly nitrogen limited, thus emphasizing the sensitivity of lower plants to nitrogen loading and potentially altering the balance between lower and higher plants.

2*.4 Mosses and liverworts as regional bio-indicators of N impacts*

Regional surveys of *Calluna* moorland have extended the work carried out at Ruabon, and have confirmed the usefulness of lichens and bryophytes as bio-indicators of nitrogen status, with clear negative correlations between species diversity and nitrogen deposition over a wide deposition gradient. Exchangeable litter nitrogen, as suggested by the rapid response in the recovery experiment, also appears to be closely related to nitrogen deposition, and could also offer a simple measure of eutrophication status.

3. Objectives

The main objective of the work to be undertaken in period 2004-2007 was to evaluate potential chemical and biological impacts of nitrogen deposition on moorland and heathland vegetation and soil systems, with detailed consideration of :

- Effects on sensitive species (mosses and lichens)
- Evaluation of the potential and rate of recovery of damaged ecosystems from nitrogen loading
- Interactions of N supply with the effects of management and differing levels of P supply
- Assessment of processes controlling nitrate leaching and leaching rates.
- Effective collaboration with researchers under the freshwater and dynamic modelling umbrellas.

These objectives were addressed over this period by continued investigations of vegetation and soil processes on the long-term nitrogen manipulation plots at Ruabon, and also by wider survey of nitrogen indicators in *Calluna* moorland. Experimental N additions and annual vegetation surveys have also been continued at the lowland heath site at Little Budworth, in order to follow the effects of the management cut carried out in Oct 2002.

4. Methods and results

4.1 Methods

4.1.1 Ruabon old Plots

The original 1m^2 plots were established in 1989 on an area of upland *Calluna* moorland in North Wales (NVC:H12) and since treated regularly with 0 – 120 kg N ha^{-1} y^{-1} as monthly applications of ammonium nitrate. Vegetation structure and *Calluna* growth have been surveyed annually. The whole area was subjected to a controlled management burn in March 2000, leading to removal of *Calluna* canopy, but leaving litter and peat layers intact. Monthly collections of soil leachate were made from both organic and mineral layers over the period 1998 – 2000.

4.1.2 Ruabon New Plots

The new Ruabon plots (4m^2) were set up in 1998 on the same and received monthly applications of nutrients site (0- 120 kg N +/- 20 kg P ha^{-1}y^{-1}) until March 2003, when the plots were split into two 2m^2 plots, with one side receiving no further treatment (termed the 'recovery' experiment. Vegetation structure and *Calluna* growth have been surveyed annually.

4.1.3 Budworth Lowland Heath

The main experimental plots at Budworth (2m^2) were established in March 1996, on an area of lowland heath in North Cheshire, and have been treated continuously (0 – 120kg N ha^{-1}y^{-1}) by monthly applications of ammonium nitrate. In the autumn of 2002 the plots were subjected to a management cut, and the vegetation removed to a height of 10 cm, and harvested. The moss and litter layers remained intact. Monthly collections of soil leachate (at 10 cm depth) were made from the plots from Dec 2000 – Oct 2004.

4.1.4 Regional survey of *Calluna* moorland

Collections of *Calluna* litter were initially made from four Peak district and five Welsh sites in a pilot study carried out in February 2004, and the samples assayed for total N and P content, and litter phosphatase activity. Two further surveys in June 2004 and April 2005 included a wider range of sites from Wales, Peak District and Scotland. Measurements included phenol oxidase activity, phosphatase activity, total N and P, KCl extractable ammonium and nitrate, and total metal content. Further surveys in late summer 2005 measured total bryophyte biodiversity, and *Calluna* and *Hypnum jutlandicum* total N on sites from Wales and Peak district. (Edmonson 2006, Caporn *et al* DEFRA 2004).

Data presented in this report is taken from a further survey of sites in Scotland, Wales and the Peak District, carried out in June 2006. The sites were carefully chosen to represent NVC: H12 late building phase *Calluna* moorland, with *Calluna* as dominant higher plant vegetation, thus allowing the collection of homogenous *Calluna* litter from under the plants.

Five locations were randomly chosen at each site, at each of which the bryophyte species in 0.25 m^2 quadrat were recorded and samples of *Calluna* litter, *Calluna* shoots, and *Hypnum*

jutlandicaum were collected for total nitrogen analysis. Measurements were also made of litter KCl exchangeable nitrogen and litter phenol oxidase activity.

4.2 Results

4.2.1 Recovery

The recovery experiment on the new plots began in May 2003, with treatment to one half of each plot halted. Data from 2004 – 2006 showed a developing pattern of consistent reductions in C*alluna* shoot extension, *Calluna* total N, moss total N and litter extractable NH_4^+ and NO_3^-. (Table 1). The clearest effects were seen at the highest rates of nitrogen input. (Figures 1 and 2), and similar effects were also obtained in 2006 in the high N + P plots. No effects were seen on *Calluna* canopy structure (height and cover), total litter and organic N, or the activity of litter enzymes.

Table 1: Recovery summary on Ruabon new plots

Component		2004	2005	2006
Calluna	Shoot extension	No effect	Significant reduction	Consistent Reduction
	Total N	No effect	Significant reduction	Not tested
Litter	NH_4^+	No effect	Significant reduction	Clear reduction at 120N
	NO_3^-	Downward trend	Downward trend	Clear reduction at 120N
Peat	NH_4^+	No effect	Downward trend	Not tested
Moss	Total N	No effect	Downward trend	Consistent Reduction
	Cover	No effect	Increased liverwort cover	

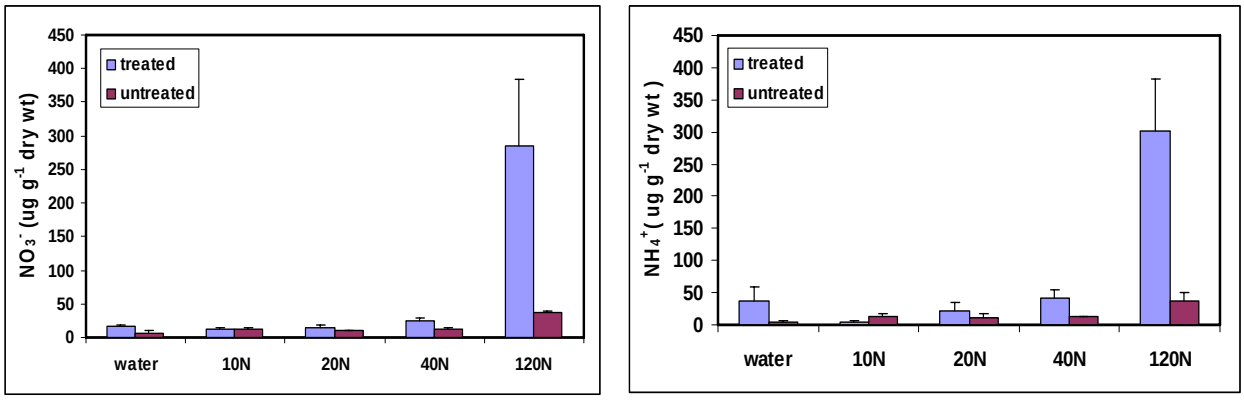

Figure 1. Litter extractable ammonium and nitrate levels. Ruabon Recovery Experiment 2006.

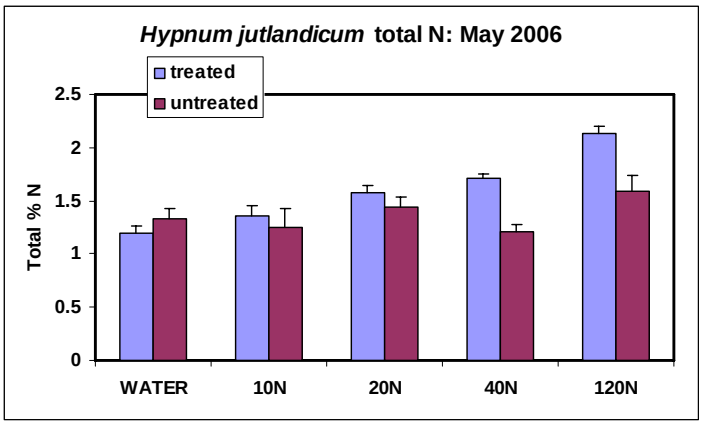

Figure 2. Moss total nitrogen content. Ruabon Recovery Experiment 2006

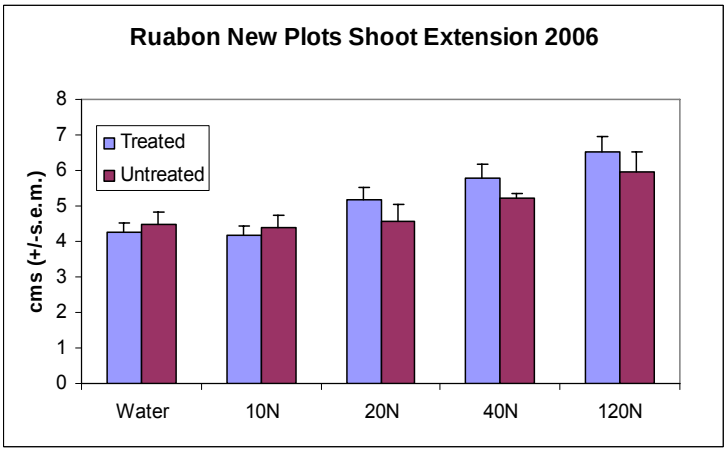

Figure 3. *Calluna* shoot extension. Ruabon Recovery Experiment 2006.

4.2.2 Management Interactions

Figures 4 and 5 show the long-term effect of natural canopy development, N treatment and management on the height of the *Calluna* canopy of the Ruabon old plots and the Budworth lowland plots respectively. At Ruabon, the gradual loss of positive response to N can be seen as the canopy ages, followed by a negative effect on canopy height following the burn. Six years after the burn the canopy is now recovering well, with a positive response to N, in terms of shoot extension (data not shown). At Budworth however, recovery on the high nitrogen plots remains very slow four years after the management cut, with very low *Calluna* cover and increased grass invasion.

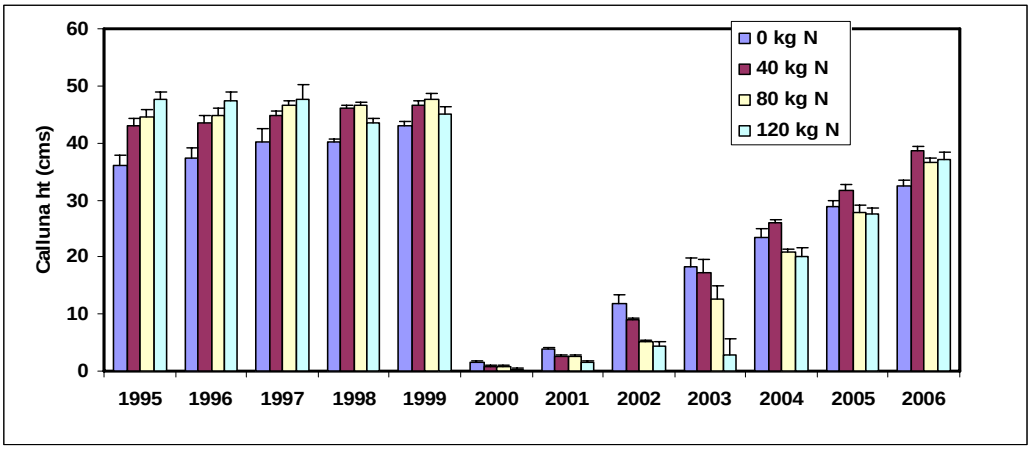

Figure 4. Ruabon Old Plots: Effects of long-term nitrogen addition and burn management in spring 2000 on *Calluna* height.

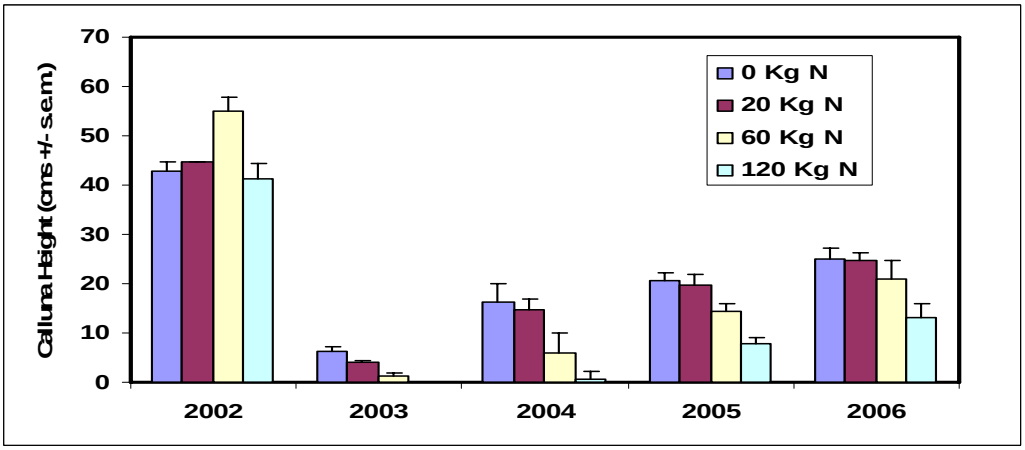

Figure 5. Budworth lowland plots: Effects of long-term nitrogen addition and cutting management in autumn 2002 on *Calluna* height.

4.2.3 Effects on lower plants

Negative effects of N on bryophyte and lichen cover were seen on both old and new plots over the period of treatments. Moss and lichen cover under the canopy of the old plots was negatively affected by nitrogen treatment at 40 kg N ha^{-1} y^{-1} and above, although some recovery in cover of certain mosses was seen in the later stages of the experiment, associated with canopy opening and increased irradiance at the highest treatment level. Nitrogen treatment also had a progressive and dose-related negative effect on moss and lichen cover on the new plots leading to significant reductions for total lichen touches at as little as 10 kg N ha^{-1} y^{-1} in 2002.

A detailed species survey in August 2005 on the Ruabon new plots found that bryophyte and lichen cover and frequency were significantly reduced by total N input, as was bryophyte species diversity (Figure 6). At a species specific scale only the moss *Campylopus flexuosus* and the liverwort *Lophozia ventricosa* were significantly reduced by total N input, although

other bryophyte species showed a downward trend. No bryophyte species responded positively to total N input.

The addition of P only or in combination with N significantly altered the negative effect of total N input on the bryophyte and lichen community. Total bryophyte cover and frequency, and the frequency of six of the 10 moss species and one of the four liverwort species all responded positively to P addition. The response of the lichens to P addition was the most pronounced, with an increase in frequency in the high N + P plot to 581% from 0% in the equivalent N-only treatment.

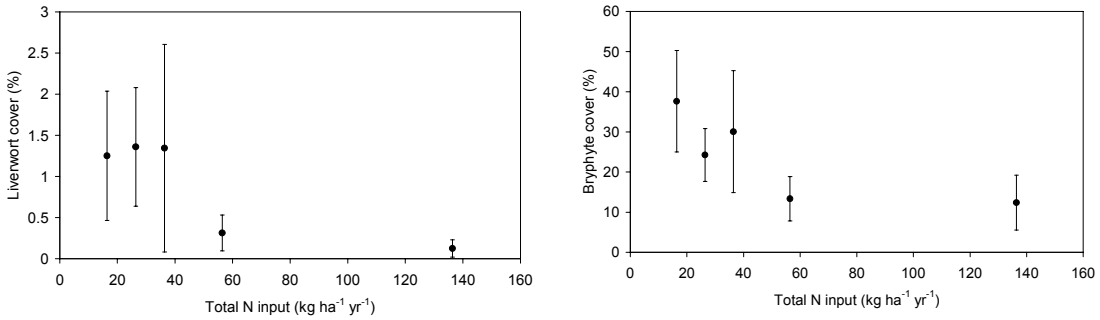

Figure 6. Effects of N treatment on bryophyte and liverwort cover. Ruabon New plots August 2005

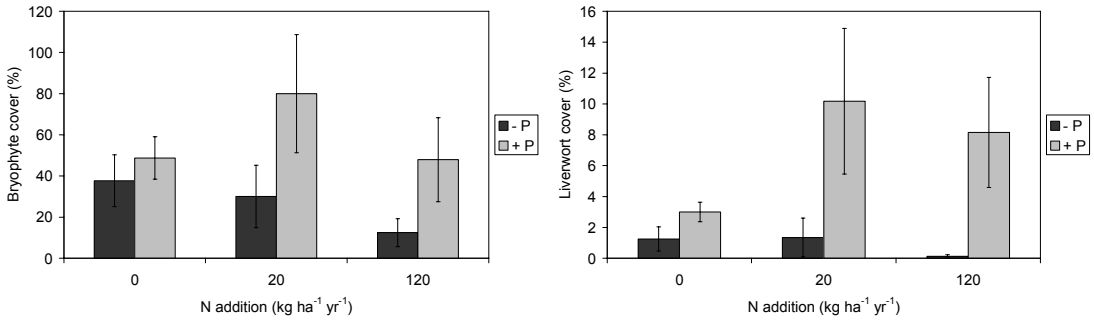

Figure 7. Effects of N and P treatment on bryophyte and liverwort cover. Ruabon New plots August 2005

4.2.4 Regional survey

The results of the regional survey of *Calluna* moorland carried out in June 2006 at sites in Wales, the Peak district and Scotland show strong relationships between a number of the chosen bio-indicators and both nitrogen deposition and atmospheric NO_2 concentration. (Table 2). The highest R^2 values were those obtained for bryophyte total species number per site, litter total N, and litter exchangeable NH_4^+. These results confirm and build on those of the earlier surveys and show the clear potential of these parameters as markers of moorland N status.

Table 2. Regional Survey of *Calluna* moorland June 2006. Linear Regression R^2 Values ($p<0.05$ *, $p<0.01$ **, $p<0.001$***.)

	Nitrogen Deposition (kg N ha^{-1} y^{-1})	NO_2 (ug m^{-3})
Bryophyte species No.	0.502 ***	0.593 ***
Litter NH_4^+	0.422 **	NS
Litter NO_3^-	0.274 *	NS
Litter-phenol-oxidase.	0.366 *	NS
Litter total N	0.700 ***	0.493 ***

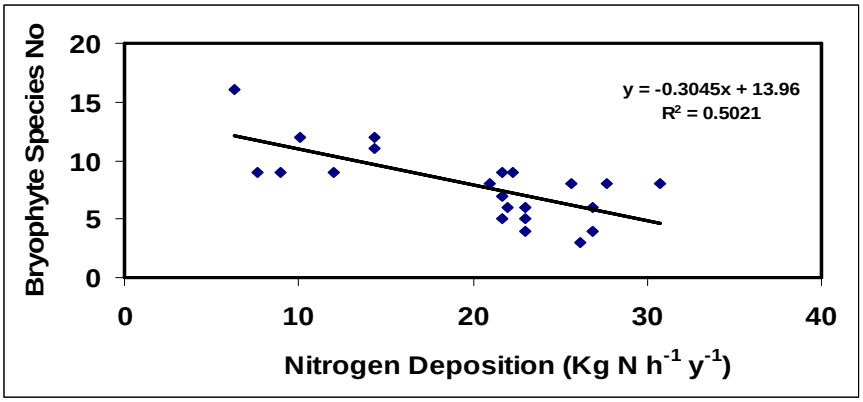

Figure 8. Relationship between Total bryophyte species number and nitrogen deposition in regional Survey 2006.

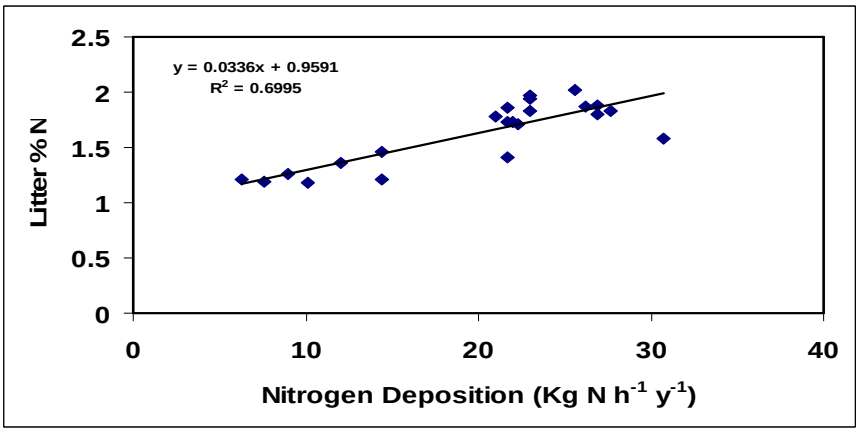

Figure 9. Regional Survey 2006: Relationship between *Calluna* litter total N and nitrogen deposition.

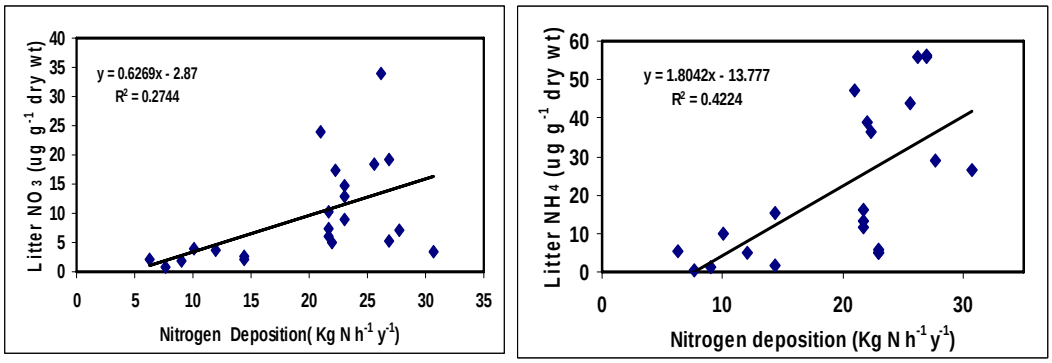

Figure 10. Relationship between *Calluna* litter extractable nitrogen and nitrogen deposition in regional Survey 2006.

5. Discussion and Implications

5.1 Recovery

Previous work on the Ruabon old moorland plots has shown that 90% of all N in the system was located in the organic soil compartment. At lower treatment levels the additional N accumulated first in the plant/litter compartments, but at higher treatment there were steep increases in the proportion of added N in underlying organic and mineral horizons (Pilkington *et al* 2005c). These changes were associated with low levels of overall nitrogen leaching and little positive growth response consistent with long-term N storage in deeper peat layers (Pilkington *et al* 2005a; Carroll *et al* 1999). The results of modeling studies additionally suggest that the accumulated N could have a positive effect on overall carbon storage (Evans *et al* 2006 a,b).On this basis, full recovery is likely to be very slow, and dependent on chemical and biological activity of the accumulated N.

Results from the recovery experiment on the Ruabon new plots however, are indicative of a pattern of reduced plant available soluble nitrogen in the litter layer, after only three years of recovery, (Edmondson 2006) suggesting that the N content of these more rapidly cycling superficial compartments may be quite responsive to current rates of input, with the potential for rapid reduction in possible leaching losses particularly of nitrate. These changes in soil chemistry have in turn resulted in significant reductions in *Calluna* growth and N content and moss total N levels, suggesting the potential for significant recovery of elements of the vegetation/litter systems over a period of five to ten years.

5.2 *Effects on lower plants*

Experimental work carried out at Ruabon over past nearly 20 years has shown a strong and consistent pattern of negative N effect on bryophyte and lichen cover and diversity on both old and new plots, at low levels of input, confirmed by detailed species level survey on new plots in 2005 (Edmondson 2006). These results can be taken as indicative of supra-optimal N loading of lower plants at near ambient deposition levels, with the potential for rapid N saturation of this sensitive section of the ecosystem as inputs increase. This conclusion in turn

receives strong support from the effect of P additions, which show clear shift to P limitation in the lower plants, under conditions in which growth of *Calluna* still appears to be N limited (Edmondson 2006, Pilkington *et al* 2007).

5.3 Regional Survey

Based on the responses to long term additions at Ruabons and results from the recovery experiment, it has therefore been possible to identify lower plants (species diversity and N contents), litter exchangeable N pools, and current *Calluna* growth as the ecosystem markers representative of rapidly cycling superficial compartments, and therefore most responsive to current deposition. The suitability of these parameters as general bio-indicators of current deposition was confirmed by the results from the regional surveys, with highly significant negative relationships between N deposition and total bryophyte species numbers, and good correlations between N deposition and litter total and exchangeable N.

There is obviously variation in both current and historical atmospheric deposition across the gradient sampled in this study, with N deposition to Wales and the Peak district lower now than 10 – 20 years ago, whereas deposition to the Scottish sites has probably always been low. Care must therefore be taken in assessing the extent to which a given indicator is measuring current deposition, or long-term "N status" or loading of a system. The distinction is likely to be dependent on the biological activity of the accumulated N, and the term "actively cycling N marker" may therefore be the most useful.

Other confounding variables, such as climatic gradients, and the current and historical SO_2 gradients also need to be taken into account, particularly in the use of the bryophytes as markers. Measurements of current bryophyte sulphur content may be useful in determining how important a factor this is in practice.

5.4 Implications of management interactions

Removal of the *Calluna* canopy by management practices such as burning or cutting form a normal part of the *Calluna* life-cycle, and should allow regeneration of the *Calluna* plants and prevent invasion by other species. They also have the effect of reducing the N load in the system, potentially leading to reduced levels of change/damage or accelerated recovery Previous results from this group involving N budgets at both Ruabon and Budworth, (Pilkington *et al* 2005c) however have shown that only a small percentage of added N (10 – 15 %) is removed by removal of the canopy.

Management manipulations at Ruabon and Budworth have shown significant negative effects of nitrogen loading in causing delayed *Calluna* regeneration following burning or cutting. This negative effect is likely to be due to accelerated ageing of N treated Calluna, and could potentially leave the system exposed to breakdown or invasion by other species. Modelling studies based on data from Budworth (Power *et al* 2004) are predictive of cyclic grass invasion following management. In practice however, even under higher N loading conditions, grass invasion has not been sustained. More frequent burning or cutting, is likely to provide the most helpful approach in nitrogen loaded systems, in addition to removing accumulated N from the system.

6. Possible future work

6.1 Long-term datasets

In order to provide data for linked soil-vegetation ecosystem model development and testing, we will collate data required for testing of linked soil-vegetation model chains from long-term manipulation sites, in particular time series data for recovery following cessation of treatments and interaction with management treatments.

6.2 Testing of indicators

To establish what are the most ecologically-relevant indicators of sensitivity to nitrogen (N) deposition and of ecosystem change due to N saturation, we will analysis experimental results in combination with new national spatial surveys to identify the most sensitive indicators for both N deposition and N enrichment.

In the case of *Calluna* moorland and related habitats, further development of the survey work is proposed to focus on the following. (i) We need to increase the number of sites, particularly in the critical load range of 10–20 kg N ha^{-1} a^{-1}, thus providing a more rigorous test of hypotheses as well as improved statistical power. (ii) A more detailed analysis is required of the relationships between nitrogen deposition and the biogeochemical and ecological status of bryophytes, including patterns of distribution and the possibility of key indicator species. (iii) More detailed investigation is needed of the factors controlling the differences in activity of phenol oxidase across the survey gradients. (iv) Investigation of other key enzymes in the decomposition process, such as b-glucosidase and the enzymes involved in nitrogen and carbon mineralisation.

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Work Package 2:
Impacts, Recovery and Processes

Task 5:
Impacts of N deposition on lowland heathlands

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Task 5 - Impacts of N deposition on lowland heathlands

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1. Summary

The long term manipulation experiment at Thursley Common has shown consistent effects of N addition on vegetation and microbial community dynamics, with associated changes in plant and soil chemistry, over the past 8 years. Earlier effects of a range of management treatments have decreased over time, although management-related differences in vegetation cover are still apparent.

A severe summer fire in July 2006 removed all of the above-ground vegetation in the experimental plots, but had only a minimal effect on the heathland soil structure. Early post-fire measurements in the ongoing N addition plots suggest that treatment-related differences in soil N availability may persist. Prior to the fire, the recovery experiment was still demonstrating the persistence of early, very low levels of N loading on the heathland vegetation, more than 10 years after the last N treatment was applied. Vegetation re-establishment and soil chemical dynamics will be monitored in both ongoing and recovery experiments to evaluate the effects of soil nutrient reservoirs and atmospheric N inputs on ecosystem recovery following the recent perturbation.

Nitrogen addition to P-limited heathland mesocosms has resulted in subtle, but distinct, effects on plant phenology, drought sensitivity and nutrient accumulation. This clearly demonstrates that P-limitation does not prevent ecosystems responding to N. A detailed nutrient budget will be determined this year, to provide a precise assessment of impacts on higher and lower plant biomass and to establish where N additions have accumulated.

A survey of more than 30 lowland heaths suggests that *Calluna* N and P concentrations are more strongly related to soil type and geology than soil chemistry. Foliar concentrations of both N and P were strongly correlated with modelled N deposition and the relationship was slightly stronger with oxidized, compared to reduced, N. The positive relationship between N deposition and both macro-nutrients clearly implies that foliar N:P ratios are not a good indicator of deposition load at a site. Data from Thursley, mesocosms and elsewhere suggest that foliar N:P ratios vary both within and between years, and that their value as indicators of relative nutrient limitation in heathland ecosystems requires further investigation.

2. Policy Relevance

The long-term experiments at Thursley Common provide evidence of the high sink capacity of nutrient-poor heathlands under modest N loads; even after eight years of elevated N inputs there was no evidence of any significant leaching of N from the system. However, one of the consequences of this ability to immobilize N is that the rate of ecosystem recovery is slow when deposition inputs are reduced. This study thus provides important insight into the speed and extent of recovery that might be anticipated as (inter)national emissions reduction policies take effect. The experiments at Thursley are unique in their longevity and low

dose/high frequency N treatments; results for both higher and lower plant performance continue to provide a strong argument for the low end of the critical load range for lowland heathlands.

Habitat management and P limitation have been suggested as potential modifiers of critical loads. Whilst more intense managements clearly remove more nutrients, the experiments at Thursley suggest that typical managements are only partly effective at reducing the effects of N accumulation. Although there are, for example, persistent effects of management on vegetation cover and soil microbial activity, effects on plant growth and chemistry were relatively transient. Furthermore, the use of P limitation as a modifier of critical loads for heathland systems is questionable, given the phonological, chemical and stress responses demonstrated in the P limited mesocosms, following N addition.

The availability of robust and reliable biological or chemical indicators of N status would provide a means of evaluating the impact of N deposition at a large number of habitats relatively quickly and simply. The lowland heathland survey has begun to assess the potential of several plant and soil chemical indicators of N status, and this will be developed further in the next phase of the programme. This approach would provide a useful, more site-specific, addition to the current critical loads approach for assessing the risk of N deposition impacts on terrestrial ecosystems.

3. Objectives

- Evaluation of the chemical and biological impacts of N deposition (1)
- Assessment of ecosystem recovery following a reduction in N inputs (2)
- Evaluation of the effects of phosphorus status and habitat management on heathland response to N deposition (3)
- Effective collaboration with researchers under the Freshwater and Dynamic Modelling Umbrellas (4)

4. Methods and results

Work carried out in the current phase of the program falls into three discrete, but related studies: The long term heathland manipulation experiment at Thursley Common (objectives 1-4), a heathland mesocosm experiment, located at Silwood Park (objective 3) and an extensive field survey of lowland heaths from south west to eastern England (objectives 1,3).

4.1 Thursley Common – long term effects of N and interactions with habitat management

The ongoing N - management experiment at Thursley Common NNR in Surrey, began in 1998; it involves addition of 0 or 30 kg N ha^{-1} yr^{-1}, to plots which were managed (in Feb. 1998) using four different techniques (low and high intensity mowing and burning). In July 2006, a severe summer fire swept through the site, burning experimental plots and resulting in the loss of all above-ground vegetation. As a result, there are no new above-ground data presented in this report for the last field season. However, prior to the fire, there had been a

consistent, significant, year on year increase in *Calluna* growth and canopy development in response to N addition. Over the previous 5 years (2001-2005), average shoot length was 60.8% greater in +N plots, average canopy height was 46.5% greater and average canopy density was 110% higher than in control plots. This increase in higher plant performance had been accompanied by a decline in understory lichen cover; before the fire, lichen cover in +N plots had been recorded at 5.6 %, compared to 53.3% in controls. Previous annual reports have also highlighted a loss of lichen diversity at this site, in response to N addition.

The consistent increase in *Calluna* biomass, together with significant increases in foliar N concentration, indicates a substantial accumulation of N in above-ground plant material since the experiment began. Over the past three years, there have been relatively few significant effects of the original (1998) management treatments; the most consistent effects have been a significantly increased cover of bryophytes (essentially *Camplyopus introflexus)* and decreased cover of *Calluna* in plots which received the high temperature burn treatment. This is in contrast to the early years after managements were carried out, where strong effects were seen for *Calluna* canopy development and grass invasion. The persistence of management effects will be re-evaluated following post-fire vegetation regeneration.

Summer fires typically occur following long, dry periods and burn at higher temperatures than autumn/spring management burns. Since the amount of N lost during a fire is a function of fire temperature (Evans & Allen, 1970), it can be assumed that the July 2006 fire removed the vast majority of above-ground and litter nutrient stores from the system. However, inspection of the soil profile immediately after the fire revealed a relatively intact, though shallow, organic layer. This suggests that soil N accumulated in the 7+ years of experimental treatments may persist. Soil samples taken in August 2006 show treatment differences in extractable ammonium concentrations (Figure 1). There was a significant effect of N treatment ($P<0.05$), of management ($P<0.05$) and sample depth ($P<0.01$). No significant treatment effects were found for extractable NO_3 or PO_4 concentrations at this time.

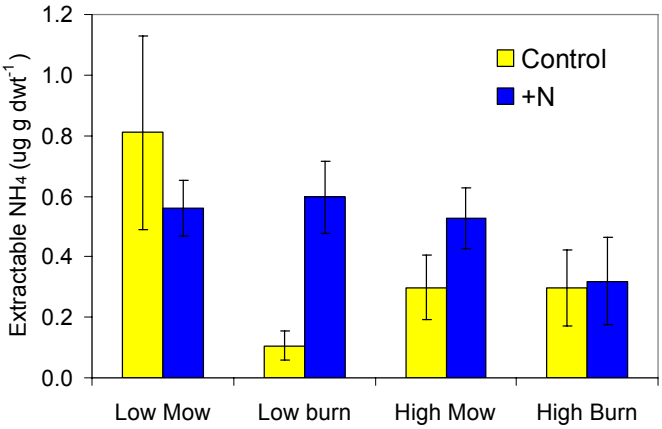

Figure 1. Extractable soil NH_4 concentrations (0-5cm) in ongoing N plots, August 2006.

4.2 Thursley Common – heathland recovery from eutrophication

The recovery experiment was set up in 1996, following 7 years' inputs of N (0, 7.7 and 15.4 kg ha^{-1} yr^{-1}). Since 1996, there have been no further treatment inputs to the plots. This study is, therefore, now in its 11th field season of recovery. Prior to the fire, there were still

persistent effects of the former N treatments apparent in terms of above-ground plant performance: *Calluna* bud burst ($P<0.01$), flowering ($P<0.01$), percent *Calluna* cover ($P<0.05$) and canopy height ($P<0.05$) were all higher in N-treated plots, compared to controls, up to 10 years following the cessation of treatments. Differences below-ground were, however, more variable. Although extractable ammonium and nitrate concentrations did not differ between treatments, soil microbial activity was still elevated in +N plots compared to controls after 8 years.

Recovery plots were also managed in February 1998 (using the same four techniques described above) and it is interesting to note that measurements of soil PME activity before the 2006 fire show a persistent effect of management ($P<0.001$); plots which had experienced the least organic matter removal (low intensity mow) had higher activities than those which had been most intensively managed (high temperature burn) (Figure 2). This indicates a relationship between P demand, enzyme activity and soil organic matter content, a phenomenon which is also apparent in the wider lowland heath survey.

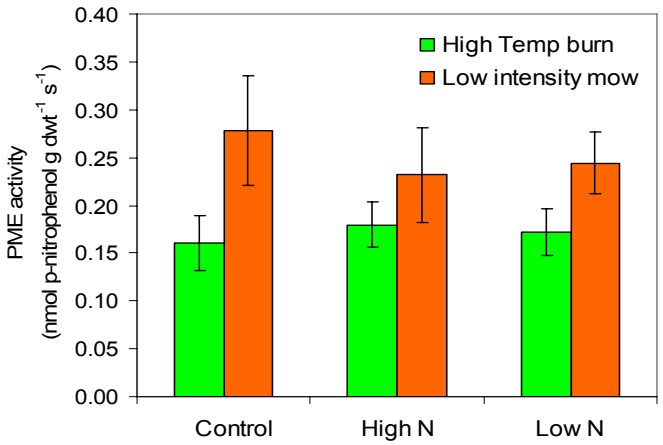

Figure 2. Effects of management type on soil PME activity in recovery plots (May 2006).

Soil extractions carried out in August 2006 show no significant differences in extractable NH_4 or NO_3 concentrations between former control and +N plots, following the previous month's fire. More detailed assessments of post-fire vegetation development and soil characteristics will be carried out in 2007/8.

4.3 Phosphorus-limited heathland mesocosms

The mesocosm experiment involves three N treatments (0 (control), 20 (N), 60 (NN) kg ha^{-1} yr^{-1}, as NH_4NO_3). These are added fortnightly, with and without the addition of NaH_2PO_4 (20 kg ha^{-1} yr^{-1}); the latter is added to demonstrate P-limitation as well as the effects of N on mesocosms when P-limitation is removed. Primary P limitation is still evident from the absence of a *Calluna* growth response to N addition, and a strong response to P addition ($P<0.001$). Interestingly, despite the lack of growth response, there is evidence of earlier *Calluna* bud burst (May 2007) in cores receiving N additions ($P<0.05$) (Figure 3).

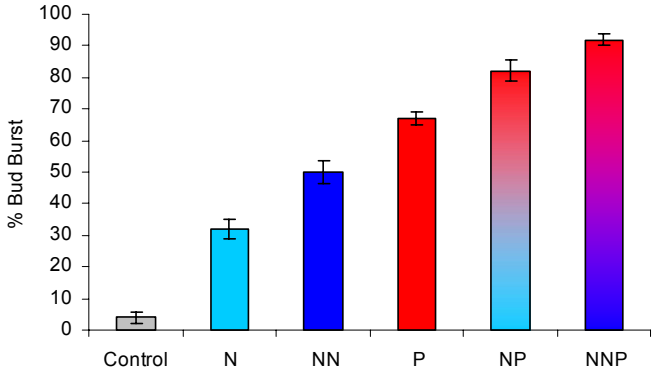

Figure 3. Effects of N (and P) treatments on mesocosm *Calluna* bud burst, May 2007.

Detailed nutrient analyses were carried out after one year and a further set of chemical, plant and microbial analyses will be undertaken in the coming months. Recent analysis of soil extractable NH_4, NO_3 and PO_4 concentrations show no significant effects of N addition, although P addition significantly decreases the amount of available NH_4, presumably as a result of increased uptake when P limitation is alleviated.

4.4 Lowland heathland survey

A survey of 33 lowland heathlands was carried out in the summer of 2005, with preliminary data analyses reported in the previous annual report. Subsequent, more detailed analyses are reported below. Samples were collected from pioneer, building and mature phase stands of *Calluna*. Concentrations of foliar N and P, and soil extractable and total N & P were determined for each phase and site, along with soil PME activities. Multivariate analysis was carried out using the "R" statistical package (v2.4.1). The aims of the survey were to: 1) quantify the variation in plant and soil chemistry across a representative range of lowland heathlands; 2) evaluate the relationships between foliar chemistry (N concentration and N:P ratios), soil nutrient characteristics and N deposition; 3) determine the effect of stand age on plant and soil nutrient status.

Foliar N concentrations ranged from 0.69% to 2.49% and N:P ratios ranged from 7.7 to 18.8 across all sites and phases. The relationships between foliar chemistry and soil nutrient availability/status were generally weak. For example, extractable NH_4 and PO_4 concentrations account for only 4.5% ($P<0.10$) and 3.8% ($P<0.10$) of the variance in foliar N concentrations, respectively. However, foliar N concentrations were significantly related to soil type ($P<0.05$) and geology ($P<0.05$) (Table 1). Areas of sand geology and/or sandy-clayey soil had lowest foliar concentrations of N (& P). Areas of mudstone geology and/or clay-loam soil had highest concentrations. Foliar N:P ratios were not significantly related to any soil nutrient indices, but there was a significant relationship with geology; the highest ratios were associated with sand or gravel, the lowest with chalk.

Table 1. Significant relationships between foliar/soil chemistry and explanatory variables in the lowland heath survey.

		Significant relationships
Foliar chemistry	%N	Total N deposition** Reduced N deposition * Oxidised N deposition* Soil type* Geology*
	%P	Total N deposition*** Oxidised N deposition*** Soil type*** Geology***
	N:P ratio	Geology*
Soil chemistry	Extractable NH_4 & NO_3	None
	Extractable PO_4	Total N deposition*
	Total soil N	PME*** Phase*
	Total soil P	PME activity* Phase*

The most significant relationship for foliar chemistry was with (modelled) N deposition (Figure 4a). Foliar N concentrations were significantly higher at sites receiving higher N inputs, and the relationship was slightly stronger with oxidized, rather than reduced, N (Figure 4b). Interestingly, the same relationship was also seen for foliar P, resulting in a lack of relationship between N:P ratios and N deposition. This is an interesting result, as it suggests strongly that N:P ratios are not a good indicator of N loading at a site. Although most studies of this type have focused on the relationship between deposition and foliar N (e.g. Pitcairn *et al*., 2001), the positive relationship with foliar P has recently been reported elsewhere (Power & Collins, 2002; Rowe, submitted).

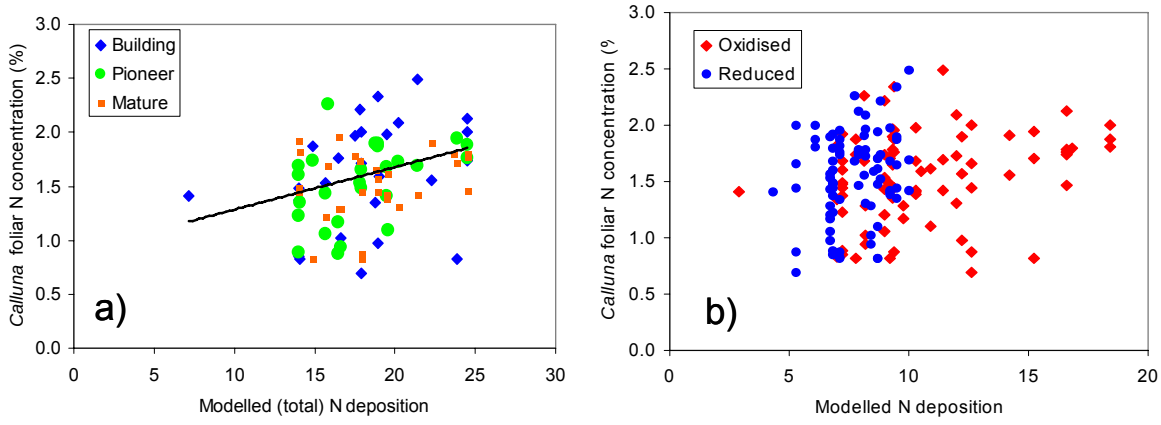

Figure 4. Relationship between a) total and b) oxidized / reduced N deposition and *Calluna* foliar N concentrations in the lowland heathland survey.

Relationships between extractable and total soil nutrients were not statistically significant. As indicated from the preliminary analyses presented previously, soil nutrient status was, however, strongly related to phase of *Calluna* growth (i.e. time since last management); both soil N and P concentrations were highest in mature phase stands, compared to pioneer/building phase.

5. Discussion and implications

The fire at Thursley in July 2006 has provided an excellent opportunity to quantify the effects of a severe, natural perturbation on nutrient dynamics at the site. Such a severe disturbance is likely to occur with increasing frequency as the climate changes. Furthermore, the fire offers the opportunity to evaluate how nutrient loading influences the rate and composition of regenerating vegetation. Initial results suggest that experimentally enhanced soil N stores may have been retained. It can be expected, therefore, that persistent treatment effects on soil nutrient availability and, potentially, also the soil seed bank will influence post-fire seedling establishment and vegetation cover. The effects of the fire on nutrient leaching are currently being quantified, and a nutrient budget will be calculated to evaluate the implications of such a disturbance for heathland nutrient dynamics, under different N loads. It will be particularly interesting to see whether this high temperature, summer fire has removed formerly persistent treatment differences in vegetation characteristics in the original (1996) recovery plots. New recovery sub-plots have been initiated, on a sub-set of the original 1998 plots; these have not received any treatment additions since July 2006. By comparing these with plots which continue to receive N treatments, the importance of soil reserves, as opposed to atmospheric inputs, for post-fire vegetation dynamics and composition can be assessed.

The heathland mesocosms have demonstrated that systems which are not primarily limited by N do, in fact, respond to N deposition. This highlights the fact that P-limitation does not fully protect heathlands from the effects of N. The mesocosms have shown either elevated foliar N concentrations, accelerated bud burst or enhanced sensitivity to drought in response to N over the past three years. This indicates that the current use of P-limitation as a suggested modifier of N critical loads may be inappropriate.

Foliar N:P ratios have been found to change in response to increasing N deposition in our manipulation experiments (both at Thursley and in the mesocosm study). However, our data suggest that the threshold ratios of <14 and >16 indicating N- and P-limitation, respectively, proposed by Koerselman & Mueleman (1996) are not appropriate for lowland heathlands. Thursley is a strongly N-limited heathland, with N:P ratios ranging from 6.6 to 17.7 over the past 5 years in control plots. Corresponding ratios in +N plots (in these same years) have consistently been slightly higher (7.5-18.3), but both control and +N plots have had ratios which would suggest P limitation in some years, despite a consistent growth response to N. Similarly, although N:P ratios of *Calluna* plants in P-limited mesocosms are lower when P-limitation is alleviated (and higher when N is added in the absence of P), ratios in the control treatment (9.1 in 2004) suggest N limitation.

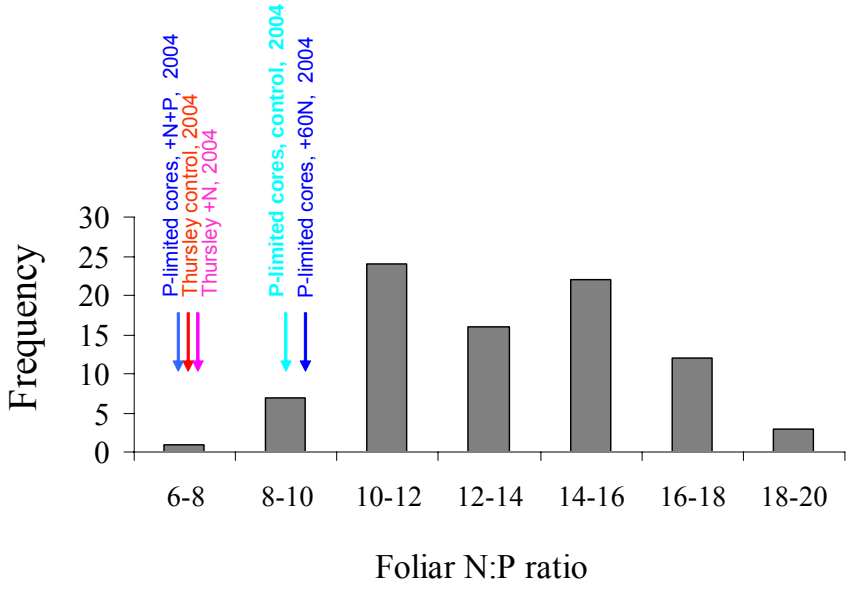

Figure 5. Frequency distribution of foliar N:P ratios for *Calluna* from the lowland heath survey, with data from Thursley (N-limited) and mesocosm (P-limited) experiments.

Figure 5 shows the frequency distribution of *Calluna* N:P ratios from the lowland heath survey, together with data from Thursley and mesocosm plants (in 2004). This highlights the fact that, despite a large range in N:P ratios recorded from the field, it is not possible to state the extent to which the lowland heathlands surveyed are N-, as opposed to P-limited. There also appears to be a considerable amount of inter-annual (and within-season) variation in foliar nutrient concentrations and ratios, which may reflect climatic variability (Green, 2005).

The relationship between foliar N (and P) concentrations and N deposition in the lowland heath survey is interesting given the wide range of geologies, climates and stand ages included. The fact that the relationship was stronger with oxidized, rather than reduced, N contrasts with the signal picked up in the CS2000 survey (Smart *et al*., 2005), but may reflect the high spatial variability of NH_x, and low range of values, compared to NO_y.

6. Possible future work

Assessment of vegetation recovery following the fire at Thursley, particularly in relation to N treatments, is a priority. This will also allow us to assess the persistence of earlier management treatment effects, as well as the relative importance of soil N stores compared to atmospheric N inputs, for regenerating vegetation. We also plan to quantify the effects of fire on nutrient dynamics, with a particular focus on N leaching. The impact of fire on ecosystem recovery from eutrophication will be assessed, in both the recovery plots which were set up in 1996 and in newly established plots. In addition, given the apparent variability in plant nutrient concentrations between years, analysis of the relationship with climatic variables will be undertaken. This will use existing (and newly generated) data from both Thursley and mesocosm experiments.

A detailed nutrient budget is needed for the heathland mesocosms, to quantify where N inputs

accumulate in a P-limited system. This will also allow us to assess treatment effects on higher and lower plant biomass more accurately than point quadrat analysis has allowed to date. Given the novelty of our findings for P-limited mesocosms, it would be sensible to assess the effects of N deposition on P-limited systems in the field. This would require setting up of at least one field manipulation at a P-limited site, but would enable us to state confidently the importance of nutrient limitation in critical load determination.

The potential value of several plant and soil chemical indicators has been assessed in the recent lowland heath survey. A more detailed survey is proposed, at a sub-set of the original sites, including a wider range of biological, chemical and physiological indicators for both lower and higher plants. Additional sites will also be included, to extend the range of N deposition covered in the survey. On-site measurements of N deposition would be advantageous, to improve the accuracy of deposition data against which to evaluate potential indicators; this would be feasible for a targeted sub-set of sites.

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Work Package 2:
Impacts, Recovery and Processes

Task 6: Interactions between grazing and nitrogen deposition at Pwllpeiran

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Task 6 - Interactions between grazing and nitrogen deposition at Pwllperian

PIs: B.A. Emmett[1], B. Griffiths[2], D. Williams[1], and B. Williams[1]

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1. Summary

The aim of this Work Package is to determine how site specific critical loads should be moderated to take into account both the form of nitrogen deposited at specific sites and the grazing intensity that site is subjected to. In addition, it is providing information on the value of some proposed indicators and providing a datasest for the development of dynamic models in Workpackage 1. The study involves a long term field experiment which manipulates grazing pressure, wet N inputs in either reduced or oxidized form at two rates and a phosphorus addition. Results indicate (i) foliar N was highly variable between years, was not affected by grazing pressure or the form N was apploed. Due to variability between years it may not be a reliable indicator of N inputs, (ii) plant reflectance (a non-destructive methodology for determining changes in physiological status of vegetation) was only successful in identifying the different grazing and phosphorus treatments but not the N treatments suggesting this method is not a substitute for direct measurement of species presence or physiological measurements which are more directly associated with N inputs; (iii) changes in soil C/N was not a sensitive indicator of N deposition or N status of a system. After 8 years of realistic N applications (10 or 20 kgN/ha/yr) no changes in soil C/N were observed, (iv) A significant reduction in the shrub species *Vaccinium myrtilus* was detected after 10 years of application of nitrogen in the form of sodium nitrate at 20kgN/ha/yr but not ammonium sulphate. In general the vegetation was more sensitive to the progressive effects of grazing pressure than nitrogen inputs with this exception where effects were of a simlar magnitude. This conclusions from this work are that increases in grazing pressure would not be effective in reducing the effect of nitrogen impacts despite an increase in N removal in sheep biomass. A reduction in grazing would benefit the recovery of some shrub and lichen species but this recovery will not be as successful where N inputs are enhanced. It could be argued that nitrogen critical loads for heavily grazed acid grassland are inappropriate due to the dominant effect of grazing pressure. No effect of nitrogen in the heavily grazed pasture was detected over the 10 years of N addition in this study. The only robust indicators of N impacts identified here or reported in past reports are direct measures of species cover and nitrate leaching rates. Soil C/N has, plant reflectance, soil gaseous fluxes, foliar %N have all found to be inconsistent.

2. Policy Relevance

Findings from two long term monitoring programmes and one spatial survey suggest that there have been wide-ranging changes in species occurrence in the UK during the latter half of 20th century associated with increased N availability. These are:

The New Plant Atlas of the UK (Preston et al. 2002) which indicated a decline in the frequency of occurrence of plant species characteristic of low nutrient availability between 1930-69 and 1987-99 and an increase in the geographic range of species associated with high nutrient availability.

The Countryside Survey (www.cs2000.org.uk) which has reported on results from repeated surveys of higher plant species data from permanent quadrats from 1978-1990 and 1990-1998. Results again suggest a shift towards plant species associated with high nutrient availability particularly in low nutrient habitats such heathland and infertile grasslands (Haines-Young et al. 2003).

A spatial survey undertaken in acid grassland by Stevens et al. (2004) in the UK which identified a decline in species richness across a N deposition gradient.

All three studies suggested N deposition as a major factor contributing to the reported shift in species composition and used various statistical approaches to support this conclusion. However, a large increase in grazing animals over the second half of the 20th century could have contributed to the shift in species particularly in infertile grasslands and heathlands. The Pwllperian nitrogen/grazing study aims to identify the differential signals of these two drivers and their interaction which will inform application of critical loads at the site specific level, identify suitable indicators, help in the interpretation of output from monitoring programmes, and assist with model development. Results have already been used to develop monitoring approaches to detect changes due to air pollution and climate change (Morecroft et al. 2005).

3. Objectives

This task aims to:

(i) identify effect of reduced and oxidized wet deposition forms of N in upland acid grassland and thus inform how site specific critical loads should be modified

(ii) identify interactions with sheep grazing and thus highlight potentially confounding signals observed in large-scale monitoring programmes

(iii) evaluate indicators of N deposition and their ecological impacts.

4. Methods and results

Nitrogen is applied fortnightly as a spray as either ammonium sulphate (AS) or sodium nitrate (SN) in deionised water to 24 replicated plots (3m x 3m) in 2 x 2ha experimental paddocks. These paddocks have contrasting sheep numbers, 1.0 sheep/ha and 1.5. sheep/ha which are termed 'low' and 'high' grazing pressure for the purposes of this study. The grazing treatments started in 1990 and nitrogen treatments in 1996. A single application of phosphorus was applied on 5th June 2000 as 20 kg P $ha^{-1}yr^{-1}$ in the form of sodium dihydrogen orthophosphate in 10 litres of deionised water to the ammonium sulphate 10 kgN/ha/yr treatment (AS10) plots to test the hypothesis that N impacts would be greater in non-P limited systems

Past measurements are wide ranging and include soil water chemistry, foliar chemistry, soil gaseous emissions, vegetation biomass and composition, net nitrogen mineralization and nitrification rates. The main objective in the first year of the project was to replace all current fencing at the site to ensure the long term security of the grazing treatments which was acheived. This was carried out including installation of stiles to prevent further damage relating to greater access by the general public. Plant reflectance was carried out in both 2005

and 2006 to match pin pointing measurements. Soil sampling was carried out in 2004 to determine changes in soil C/N status. Vegetation composition assessment using the pin pointing approach was planned for 2004 but was postponed to 2005 due to problems finding suitable recorders and assessment was repeated again in 2006. Results are reported in 3 main categories: (i) testing of indicators of both grazing and N addition, (ii) response of vegetation composition assessed, (iii) change in soil C and N status which is an important variable for the initiation of the dynamic models under development.

4.1 Testing of indicators

Some key new findings are:

(i) No evidence of grazing intensity effects on foliar % N chemistry for a wide rage of plant species. It should be noted that ungrazed systems which are not included in this study may respond differently (Figure 1.). This suggests that grazing pressure will not confound the use of foliar %N as an indicator of N deposition in these systems.

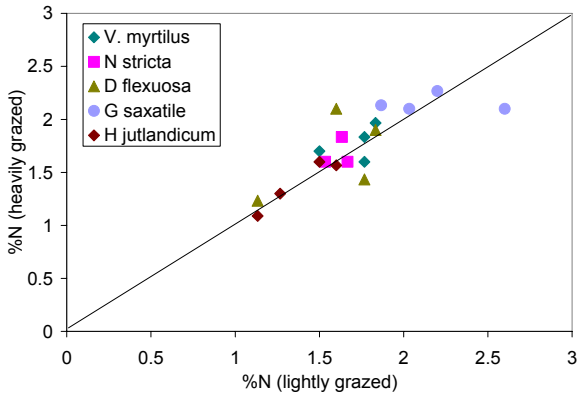


Figure 1. The relationship between %N sampled over four consecutive years in acid grassland with light versus heavy grazing. A 1:1 line is shown.

No difference in the %N in ammonium sulphate and sodium nitrate treated plots were observed indicating that there was no consistent effect of N form in wet deposition on foliar N response if N is applied at realistic low doses.

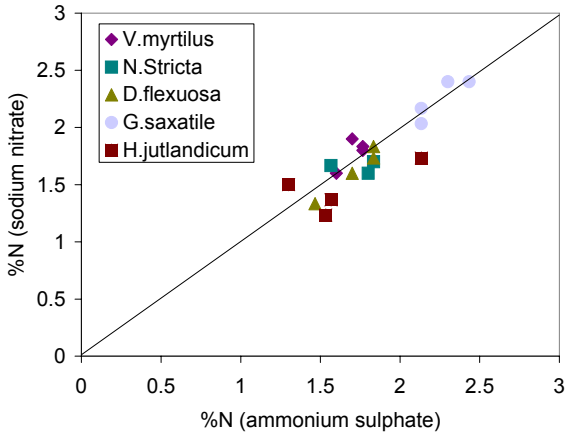


Figure 2. The relationship between %N sampled over four consecutive years in ammonium sulphate 20 kgN/ha/yr treatment plots and sodium nitrate treatment plots 20 kgN/ha/yr. A 1:1 line is shown.

(ii) Significant inter-annual variation in foliar %N concentrations in many species (Figure 3.). This suggests trends in %N in the short term should be treated with caution as an indicator of changes in N deposition as climatic factors clearly have a large effect on foliar nitrogen content.

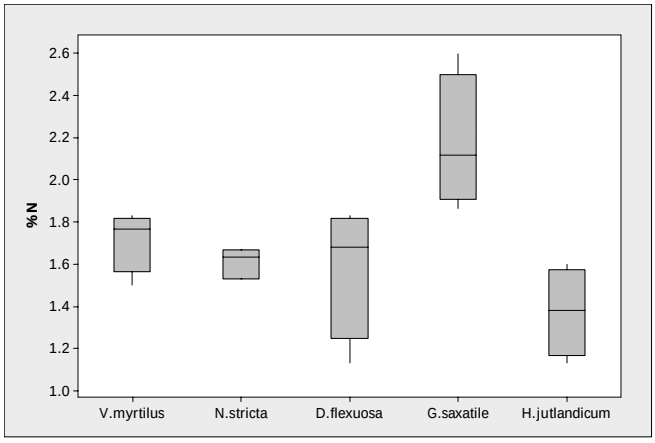


Figure 3. The variability in %N of different species over a four year period in control plots in an acid grassland.

(iii) Spectral radiometry is a measure of the radiance of light from the vegetation canopy of wavelengths between 304 - 1135nm measured at a height of 50cm above the vegetation canopy. It is used as a non-destructive approach to assessing the physiological status of plants (Penuelas and Filella 1998). It forms the basis of remote sensing. As it has not been described before in previous reports, details of the approach are presented here.

Measurements are referenced to an ambient light reading using a cosine receptor which in turn is referenced to an ambient light reading taken above a white standard minus the dark scan value (this eliminates any sensor noise from the measurement and gives a standard to work from) Several relationships can be derived from the radiance values by simple calculation, these are:

- Water index (WI): This is a measure of the water content within the vegetation canopy, calculated by R900/R970nm. (The higher the number the greater the water content of the vegetation canopy).

- Simple Ratio (SR): This is a measure of the Green Biomass within the vegetation canopy, calculated by R800/R680nm (The higher the number the greater the amount of Green Biomass)

- Normalised Difference Vegetation Index (NDVI): This again is a measure of the Green Biomass of the vegetation canopy , this time calculated by (R800-R680)/(R800+R680)nm (The higher the number the greater the amount of Green Biomass but is not as sensitive as the Standard Ratio)

- Structural Independent Pigment Index (S.I.P.I.): This is a ratio of Caratenoids to Chlorophyll a and is related to plant stress, calculated by (R800-R445)/(R800-R680)nm (The higher the number the greater the stress)

- Photochemical Reflectance Index (P.R.I.): This is a measure of the photon use efficiency within the vegetation canopy and is related to photosynthesis, calculated by (R565-R532)/(R532+R565)nm. (The higher the PRI number, the lower the ratio of carotenoids to chlorophyll a)

- Total Chlorophyll: This is an empirical value for the total of Chl a and Chl b calculated by (0.0202*R645)+(0.00802*R663)*1000 (The higher the number the greater the Chlorophyll concentration which is closely correlated to nitrogen status of the vegetation)

There are no standard units associated with any of the above indices, all values are empirical. We made measurements over the plots on 13th May 2005 and 16th July 2006. The instrument sensor was held over individual point quadrats at 0.6m height which reads a 0.5m diameter circle and 4 measurements were made per plot. Mean of these were taken and standard error reflects variance between these means.

Effects are variable between the two sets of measurements due to different times of year (spring in 2005 and summer in 2006) and inter-annual climate differences and thus results are discussed separately:

Spring:

- No significant effect of N treatments (AS20 or SN20) was observed for any index.

- No differences in green biomass, chlorophyll or photosynthestic efficiency were observed between the 2 grazing paddocks but greater water stress was observed in the heavily grazed paddock. (Figure 4.).

- A large positive response to phosphorus ($P < 0.05$) was observed with significantly greater green biomass as indicated by the NDVI and SR indices in both grazing treatments. Significantly lower SIPI value suggest lower stress of the plants but significantly PRI values also indicate a lower photosynthetic efficiency. This is probably linked to the lower chlorophyll content detected suggesting an ‘effective’ reduction in N status of the vegetation in the phosphorus plots

Summer:

- A trend towards reduced green biomass (as indicated by NDVI and SR) due to higher animal numbers in the high grazing paddock was apparent in mid-summer as expected. No effect on chlorophyll (and thus N status) was detected. Grazing also induced greater stress (as indicated by SIPI index) and a lower photosynthetic efficiency (PIRI) in heavily grazed paddocks. (Figure 5.)

- Phosphorus again significantly increased green biomass ($P < 0.05$) and trends for reduced stress levels (not significant) but only in the heavily grazed paddock. No effect on other indices was observed.

- No effect of N treatment was observed relative to the controls.

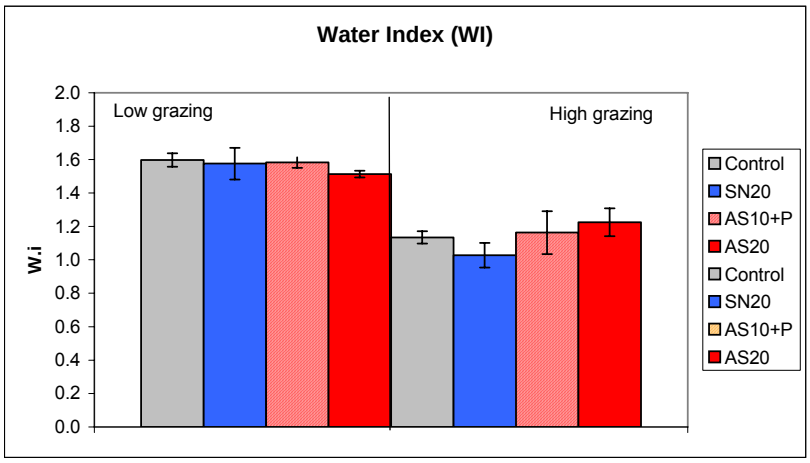


Figure 4. Effect of grazing intensity on water status of vegetation in spring 2005. No effect of grazing was observed in summer 2006. (AS10 = ammonium sulphate 10 kgN/ha/yr + phosphorus; AS20 = ammonium sulphate 20kgN/ha/yr; SN20 = sodium nitrate 20 kgN/ha/yr.)

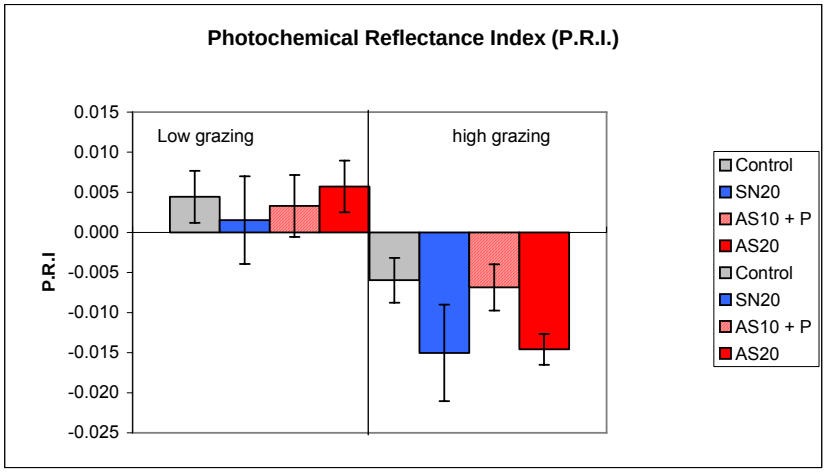


Figure 5. Effect of grazing intensity, nitrogen and phosphorus on photosynthetic efficiency of vegetation in summer 2006. (AS10 = ammonium sulphate 10 kgN/ha/yr + phosphorus; AS20 = ammonium sulphate 20kgN/ha/yr; SN20 = sodium nitrate 20 kgN/ha/yr.)

4.2 Change in vegetation composition

Key findings after 10 years of N applications and differential grazing treatment included:

Results were analysed using a General Linear Model for each paddock separately using cover recorded in 1996 prior to the start of treatments as covariate. If significance was detected, differences between treatments was tested using Tukey's test. Significance was accepted at P 0.05 level.

Cover of *Vaccinium myrtilus* declined over time suggesting the original ESA treatments were too high to promote recovery of a shrub community (Figure 6a).

A significant reduction in *Vaccinium myrtilus* with sodium nitrate 20 kgN/ha/yr treatment under low grazing. No change was observed in the ammonium sulphate 20 kgN/ha/yr (Table 1). N treatements had a similar effect to the grazing pressure on cover over the 10 year period.

The decline in Vaccinium myrtilus was related to an increase in Carex pilufera (Figure 6b, Figure 7 a & b)) in the sodium nitrate treatments although changes in Carex were not significant in themselves.

There was a significant reduction in the cover of the rank grass *Nardus stricta* in response to 20 kgN/ha/yr applied as either ammonium sulphate or sodium nitrate probably due to the enhanced quality of foliar material and associated higher grazing pressure. (Figure 6b)

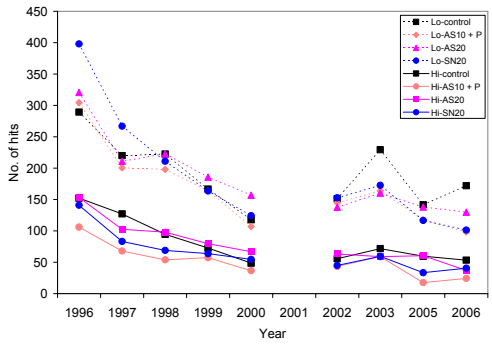
Other trends noted (not significant) are listed in Table 1 and Figures 8 a – d. As can be seen it is difficult to make any general statements concerning the effect of N form or phosphorus. Responses are highly species dependent. In summary, vegetation cover has changed over time due to the original ESA prescriptions for grazing which has encouraged the increase

cover of the rush *Juncus squarrosus* but not as expected *Nardus stricta*. The mosses *Hypnum jutlandicum* and *Rytidiadelphus loreus* declined with this grazing pressure although this was offset by the addition of phosphorus in the latter case.

Table 1 Summary of response to nitrogen form, grazing pressure, their interaction and phosphorus. Trends shown with significant response identified at significance level in shaded cell.

Species	**Time**	**Low grazing**			**High grazing**		
		Ammonium sulphate	**Sodium nitrate**	**Phosphorus**	**Ammonium sulphate**	**Sodium nitrate**	**Phosphorus**
Vaccinium myrtilus	-	-	- P < 0.05	-	=	=	=
Nardus stricta	-	- P < 0.001	-	-	- (p < 005)	- (P< 0.05)	- (P < 0.001)
Carex pilufera	=	+	+	=	=	=	=
Deschampsia flexuosa	-	=	=	=	=	=	=
Festuca ovina	=	=	=	=	=	=	=
Galium saxatile	-	-	=	+	-	=	+
Juncus squarrosus	+	-	-	+	-	-	-
Potentilla erecta	=	I/S	I/S	I/S	I/S	I/S	I/S
Rhytidiadelphus loreus	=	-	=	+	-	-	+
Rhytidiadelphus squarrosus	-	I/S	I/S	+	I/S	I/S	I/S
Hypnum cupressiforme	-	I/S	I/S	I/S	=	=	=
Pleurozium schreberi	=	-	-	+	-	-	+
Racomitrium lanuginosum	=	=	=	=	=	=	=
Lichens	-	-	-	=	I/S	I/S	I/S

a)



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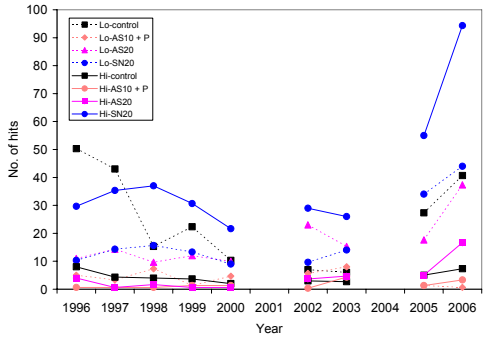

Figure 6 Change in combined number of hits at 1cm, 5cm, 10cm and max height for; (a) *Vaccinium myrtilus*, (b) *Carex pilufera*

a)

b)

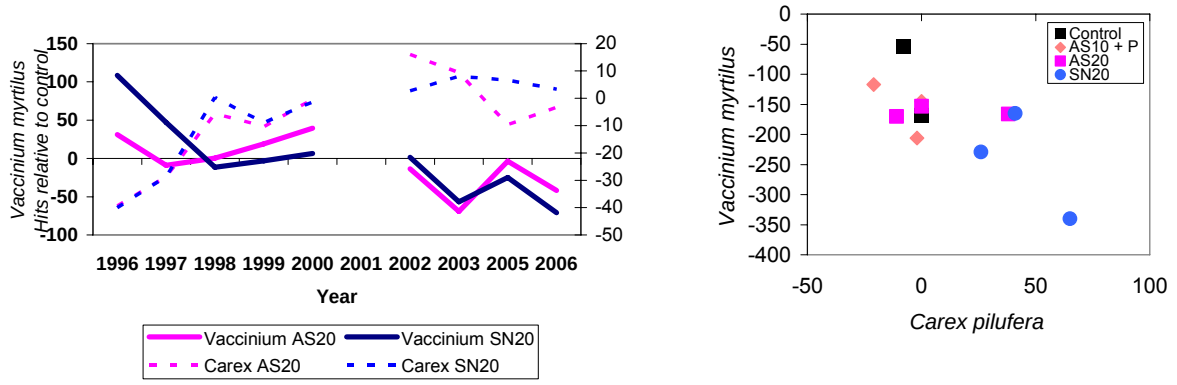

Figure 7 The relationship between change in *Vaccinium myrtilus* and *Carex pilufera* between 1996 and 2006 (a) over time (b) between 1996 and 2006

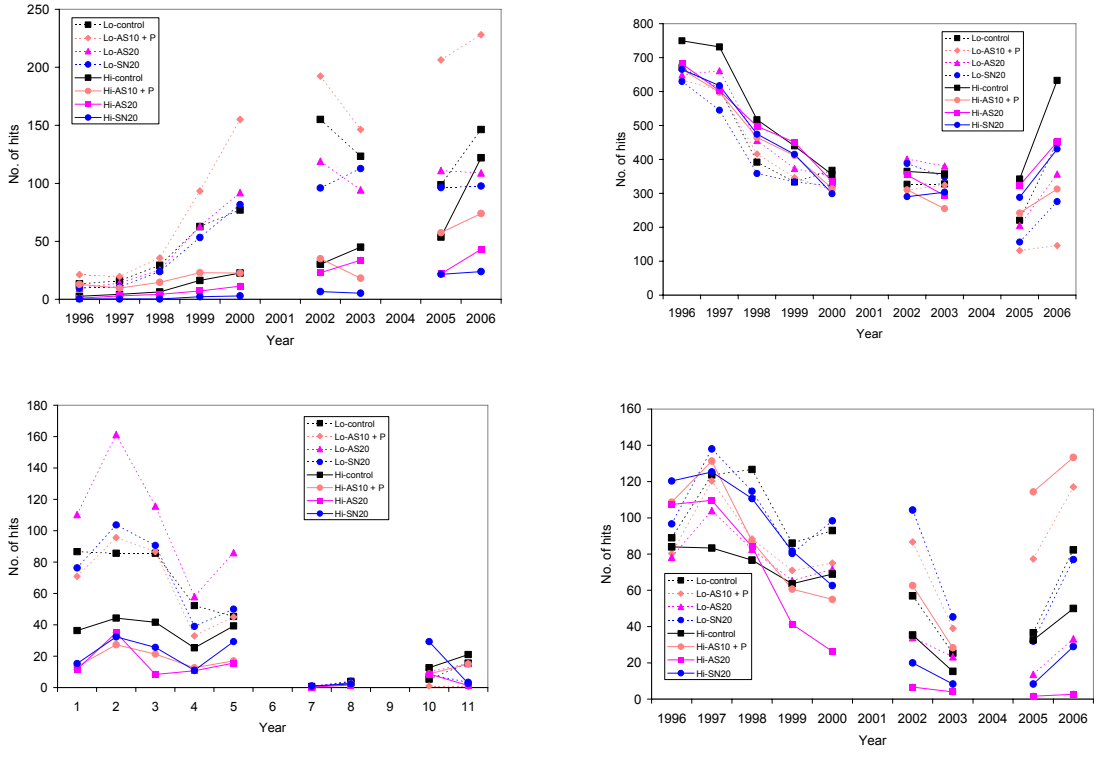

Figure 8 Change in combined number of hits at 1cm, 5cm, 10cm and max height for; (a) *Juncus squarrosus*, (b) *Nardus stricta, (c) Hypnum jutlandicum (d) Rhytidiadelphus loreus*

4.3 Change in soil N status

Resampling of the soils was carried out all 24 plots in 2004. No significant differences ($P < 0.05$) in %N, C/N of organic layer or 2 lower mineral horizons or the mass of the organic layer were observed (Figure 7a and b).

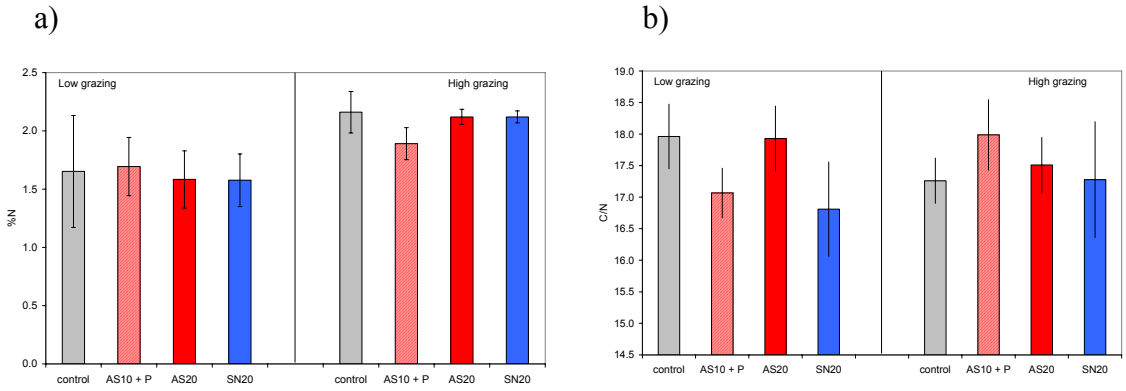

Figure 7. Soil C and N status in the different treatment plots: (a) %N in organic layer and (b) C/N in organic layer. Standard error bars are shown.

It is often difficult to detect changes in soil N and C stocks due to the large pools already present in the soil. At the low levels of application (10 and 20 kgN/ha/yr) at this site compared to some of the other experimental sites, it is perhaps not unsurprising that no change is detected.

5. Discussion and implications

Results indicate that grazing pressure at this site has been the most important driver of change to vegetation composition. In particular a decline in the shrub species *Vaccinium myrtilus.* This decline was accelerated by the addition of nitrogen in the form of sodium nitrate which was associated with an increase in *Carex pilufera* perhaps due to competition.

Responses to nitrogen, phosphorus and grazing was highly specific and no generalizations could be made regarding the risk of change associated with N form.

Change in soil C/N was not found to be an early signal of change in vegetation composition.

Our hypothesis that the use of plant reflectance could provide new information to help understand underlying mechanisms for changes in vegetation composition in response to N, P and grazing treatments e.g. changes in water stress. Differences due to grazing were detected primarily related to increased water stress which has implications for productivity response to climate change. In addition, the 'greening' of the vegetation due to phosphorus addition was also successfully detected. Variability between plots resulted in no detection of N treatments indicating baseline measurements would probably be required for use in N impacts work.

A synthesis of the findings from this study were presented at the Biogeomon 5th International Symposium on Ecosystem Behaviour in Santa Cruz, California, June 2006. This work also

contributed to the scoping study to develop monitoring approaches for air pollution and climate change (Morecroft et al. 2005).

6. Possible future work

Future work should focus on obtaining one full year of vegetation, water, gaseous and soil data for one full year after long term additions of N, grazing and P to test soil-vegetation model chains under development. A current full dataset is available for 1996 – 1999 with vegetation data only available from 1996 – 2003, soil data for 2004 and vegetation data only again in 2005 and 2006.

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Work Package 2:
Impacts, Recovery and Processes

Task 7:
Whim moss N manipulation experiment and open-top-chamber flux work

L. Sheppard, I. Leith, A. Crossley, N. van Dijk
M. Sutton, N. Cape, D. Fowler, M. Jones
M. Prendergast, I. De Lange, E. van Zetten
P. d' Hooghe, H. Holding and C. Field

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Task 7 - Whim moss N manipulation experiment and open-top-chamber flux work

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1. Summary

- The Whim bog N manipulation experiment, treating an ombrotrophic bog with real time simulations of gaseous ammonia emissions from an agricultural point source, and with N enhanced wet deposition, to compare effects of gaseous ammonia (NH_3), wet deposited oxidised N (nitrate, NO_3^-) and wet deposited reduced N(ammonium, NH_4^+) has run uninterrupted for 5 years (June 2002-2007).
- A flux chamber study was undertaken, (as a PhD) to characterise the relationship between NH_3 concentration and N deposition for different bog species.
- Ammonia, in concentrations typical of air in the immediate vicinity (<100 m) of agricultural point sources has been shown to damage sensitive bog/moorland vegetation.
- All plants are not equally sensitive to ammonia. Many of the semi-natural species growing on the bog appear to be highly tolerant *e.g. Erica tetralix*, *Vaccinium myrtillus* and *Empetrum nigrum*.
- Negative effects of ammonia are species-specific, with sensitivity related to characteristics that influence the deposition, uptake and assimilation of ammonia, and the consequent effects of ammonia on species' ability to deal with other stress whether abiotic *e.g.* drought and biotic *e.g.* pests and pathogens. *Cladonia portentosa* is more sensitive than heather (*Calluna*) which is more sensitive than bog moss, *Sphagnum capillifoliu.*
- Long-term exposure to ammonia causes plants to be damaged by lower ammonia concentrations *ie.* the damaging effect over time is cumulative.
- Results from the Whim bog experiment were central to the setting of a new Critical Level for ammonia of 1 μg m^{-3} (cf. previous value of 8 μg m^{-3}) for ecosystems containing lichens and bryophytes. This has now been accepted by the relevant parties within the UNECE and is awaiting formal ratification in December 2007.
- Intermittent high NH_3 concentrations close to point sources, concealed within the time averaged values, are probably responsible for the observed damage.
- Ammonia deposition is concentration dependent, species specific and varies according to the roughness of the vegetation canopy and species assimilation capacity.
- The use of ammonia concentrations rather than equivalent N deposition may be a more useful tool for environmental regulators to operate with, since concentrations

can be relatively easily and cheaply monitored, and the calculation of equivalent N deposition, from concentrations leads to large uncertainties.

- Ammonia is far more effective at elevating foliar N concentrations and causing biochemical changes, which may be evidence of phytotoxicity, than the equivalent wet deposited N load.
- Enhanced N deposition accelerates aging in *Calluna* and thus its ability to regenerate. However, recolonisation by seedlings has not so far been impaired, even though mineral N is measurable in the soil water (an indication of N saturation).
- Responses of foliar N to enhanced N deposition vary with N dose, N form and plant species. Moss N status responds significantly more to the N dose than *Calluna* N status, (greater dilution potential relative to uptake area).
- Five years of wet deposition of NH_4^+ and NO_3^- have not so far, caused species loss, and visible damage has been minimal, even though the highest experimental N addition (56 kg N $ha^{-1}a^{-1}$) is at the maximum end of the range of UK wet N deposition.
- Environmental factors such as climate, especially drought and the availability of other limiting nutrients *e.g.* phosphorus and potassium, can strongly influence the impacts of N.
- Realistic simulations of N deposition, in this experiment, to an ecosystem that was previously relatively N 'clean' in comparison with other UK experimental sites, have shown that previous N deposition history affects the vegetation response and should be factored in to Critical Load calculations.
- Addition of P+K in combination with N appears to significantly increase the tolerance of the lichen population to wet deposited N, promoting colonisation by nitrophytic, epiphytic lichens *e.g. Xanthoria parietina* and *Physcia tenella*, but now is having a less positive effect on *S. capillifolium* than reported in Carfrae et al. (2007). Wet oxidised N with P+K at the highest dose (56 kg N $ha^{-1}a^{-1}$) significantly increased the cover of the pleurocarpous moss *Hypnum jutlandicum* at the expense of *Calluna*. The combination of reduced N with P+K did not promote increased cover of *Hypnum*.
- Wet deposited ammonium (reduced N) led to acidification of this acid peat soil but wet deposition of nitrate (oxidised N) at high doses and dry deposition of ammonia gas have increased soil and soil water pH.
- The different effects of reduced versus oxidised N observed so far: acidity (ammonium increased soil acidity whereas nitrate decreased soil acidity; effects on moss base cations (ammonium lowered base cation concentrations in *Hypnum* and *Sphagnum* > 20% whereas nitrate had no significant effect; nitrate probably increases growth rates) are not after 5 years conclusive, as to whether wet ammonium is more detrimental for ombrotrophic bog ecosystems than nitrate N.
- Enrichment of the available mineral N pools has increased, but the high variability means that a minimum of one year of data is needed to establish trends that can be used to fully evaluate the Critical N Load for ombrotrophic bogs.

2. Policy Relevance

The unique field experiment which is treating an ombrotrophic bog with real time simulations of dry deposited NH_3 and wet N deposition, to compare effects of gaseous ammonia (NH_3), wet deposited oxidised (nitrate, NO_3^-) and wet deposited reduced N (ammonium, NH_4^+) has addressed:

1. The importance of the type of N deposition, (dry versus wet) and the significance of the N form, (reduced versus oxidised N).

2. The debate surrounding the Critical Level versus the Critical Load for ammonia. The flux chamber study has addressed the issue of the relationship between ammonia concentration and N deposition.

3. Indicators of N effects and whether they are fit for purpose.

4. Consequences of N deposition on the biogeochemical cycling of N and C on an ombrotrophic bog.

5. The role of other nutrients (phosphorus and potassium) in moderating the effects of wet N deposition.

2.1 Field Evidence for ammonia effects

We have the first field evidence, for ombrotrophic bogs, that three N forms affect plant and soil responses differently, with respect to both the type and scale of effect. Dry deposited ammonia, reduced N, in concentrations and conditions simulating those measured close to intensive livestock units, can be phytotoxic. This ***in situ*** experiment on a natural ecosystem, with negligible disturbance of the site and minimal infrastructure, has shown that ammonia damages a range of plant functional groups; *Calluna,* an evergreen shrub (ericoid), the mat forming lichen, *Cladonia portentosa* and the keystone bog moss, *Sphagnum capillifolium,* and that the effects are strongly environmentally mediated, *e.g*. exacerbated by drought. The prolonged nature of the experiment, 5 years, has revealed the significance of long-term exposure to ammonia *cf.* Critical Loads; sensitive plants exposed to NH_3 for several years have a lower concentration damage threshold. Visible damage produced under these experimental conditions, where cause and effect can be attributed with confidence, appears to be symptomatic for NH_3 exposure. These visual symptoms can thus be used to evaluate whether damage observed near point sources is likely to have been caused by ammonia exposure.

The Whim experimental results were central to the discussions at the recent **UNECE ammonia critical levels workshop in Edinburgh** where a proposal was made to revise the ammonia critical levels (CL_{NH3}) to take account of the cumulative effects of exposure to NH_3.

2.2 Flux chamber: NH_3 concentration, deposition and species sensitivity

Ammonia-N deposition was shown to be: concentration dependent (higher at lower concentrations and lower at high concentrations which can saturate the system, leading to re-emission), species specific and influenced by light and temperature (through their effects on

N assimilation), affected by the presence of stomata, wind speed (via effects on aerodynamic resistances) and surface wetness, increasing when the wet surface is acidic (Jones, 2006; Jones et al. 2007). The absence of linearity between concentration and deposition means that many past studies have under and over estimated NH_3-N deposition (Jones et al. 2007). Bog plants such as *S. capillifolium* and *Cladonia* spp. represent large sinks for NH_3, due to their large, often wet and acidic surface area. The large potential for uptake together with an inherent lack of control over uptake and relatively low assimilation capacities makes such species highly vulnerable to ammonia exposure. Given the complexities and uncertainties identified in calculating NH_3-N deposition for different ecosystems compared with measurement of NH_3 concentrations using passive samplers we suggest that NH_3 concentration may be a more useful tool for the environment agencies to use in emission ceiling enforcement.

2.3 Field evidence for wet effects, reduced versus oxidised N

Nitrogen supplied as either wet reduced or wet oxidised N, up to 56 kg N $ha^{-1}a^{-1}$, has not after 5 years, caused obvious damage compared with that caused by ammonia. However, subtle changes (indicators) have been recorded, above and below ground, which may become more important with time. These observations show that factors such as N deposition history, (relatively N clean site, ambient N deposition ~ 10 kg N $ha^{-1}a^{1}$), vegetation type (NVC M19 *Calluna vulgaris-Eriophorum vaginatum* blanket bog with aging *Calluna*, unmanaged) and method of treatment (high frequency, low concentration coupled to rainfall, no rain = no treatment) may influence ecosystem response to wet deposition. Our results suggest that less natural field N manipulation studies may have exaggerated plant responses to N. The experiment, to date, has provided no clear evidence to support the hypothesis that wet reduced N is more damaging for bog species than wet oxidised N, although the two forms have caused several contrasting effects. Where the different N forms have been provided with P+K in proportion to N however, there has been a large positive effect of high oxidised N on the cover of pleurocarpous moss and a negative effect on *Calluna* cover.

3. Objectives

- Evaluation of the potential chemical and biological impacts of N deposition.
- Assessment of the role of wet versus dry N deposition in contributing to eutrophication.
- Assessment of the importance or reduced versus oxidised N.
- Effective collaboration with researchers under the Freshwater and Dynamic modelling Umbrellas.

4. Methods and results

Figure 1. Aerial view of Whim bog nitrogen manipulation study site, Peeblesshire, Scottish Borders.

The application of the different N forms in this field N manipulation on Whim bog (see Figure 1) is globally unique. The vegetation represents a repeating mosaic dominated by *Calluna*, the sedge *Eriophorum vaginatum* and bog moss *Sphagnum capillifolium*. Three N forms are being compared on the one site, and the N applications are determined by and coupled to meteorology, to provide real time simulation of wet and dry deposition. The wet and dry N treatments are provided all year round, when rainfall availability, wind speed and wind direction permit (Sheppard et al. 2004; Leith et al. 2004). Wet treatments provide 8, 24 and 56 kg N $ha^{-1}a^{-1}$ above the ambient N deposition (8-11 kg N $ha^{-1}a^{1}$), at concentrations < 4mM in rainfall collected on site, on ~120 occasions per year, providing ~ 10% additional precipitation to treated plots. Reduced N is provided as NH_4Cl and oxidised N as $NaNO_3$ to four 13 m^2 replicate plots, from a 360^o central sprayer in each plot. The high and low N doses are also provided with P and K in a 1:14 P:N ratio as K_2HPO_4. The dry NH_3 gas is mixed with air and released from a 10 m line source, 0.5 m off the ground, for 3-14% of each month, depending on wind direction. Ammonia concentrations are monitored along the transect, 0.1m above the vegetation canopy, (Figure 2) using ALPHA samplers (Tang et al 2003) changed monthly. Meteorological conditions are logged every minute, verified and meaned to provide daily averages, or accumulated for rainfall. NH_3-N deposition (Figure 2) is calculated using the revised equations for this M19 vegetation, based on flux chamber measurements, and taking into account NH_3 dependent R_c, actual NH_3 concentration, solar radiation and wind speed, for the Whim mixed bog community (Jones et al. 2007).

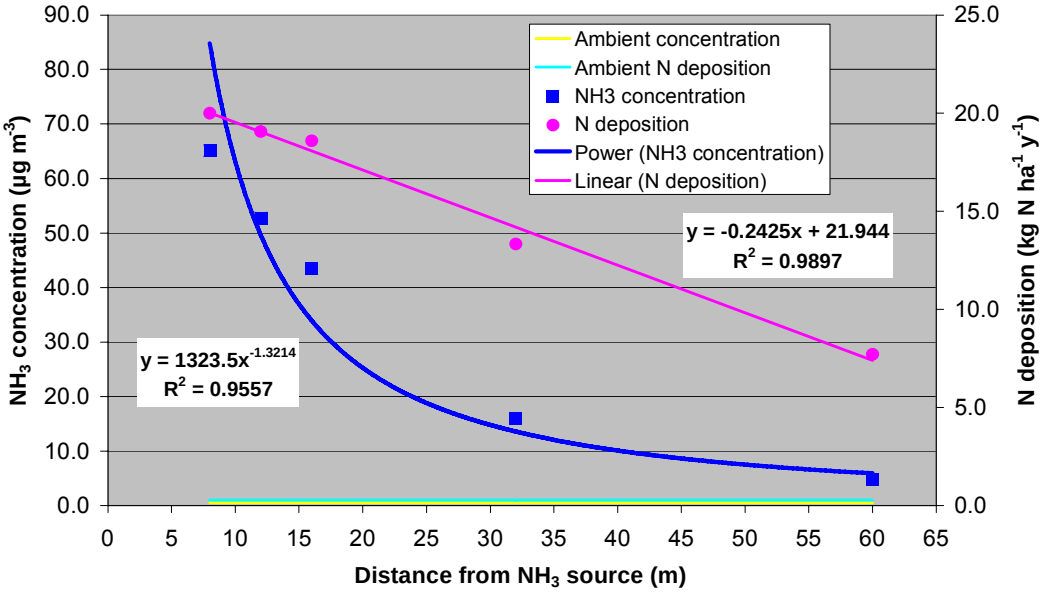

Figure 2. Ammonia concentration (measured at 0.1 m above the vegetation) and NH_3-N deposition (in 2004) along the ammonia release transect.

Species cover change is recorded in three 30 cm^2 quadrats per plot, each divided into 16 squares and 3 plots per distance from the NH_3 source. A weighted cover index based on the pretreatment cover and the proportion of the square covered was calculated ((initial cover-cover year measured)/average(start + cover year measured/2))for the main species and categories *e.g.* live or dead. *Calluna* growth has been measured retrospectively from October 2006. Foliar nutrients have been measured with a CN analyser and in a standard acid digest with Se catalyst for P and base cations, using ICP-OES. The pH of the different species was measured after macerating in deionised water, using a ball mill.

Soil water is collected monthly at 5-10 cm below the litter layer using rhizon samplers with a 0.45 µm filter and analysed for DOC (Thermalux C analyser), Na^+, Ca^{2+}, K^+, Mg^{2+}, NH_4^+, NO_3^-, PO_4^-, Cl^- and SO_4^{2-} by IC. Extractable mineral N is monitored quarterly, as is soil mineralization. NH_4^+ and NO_3^- are extracted in 0.9 M KCl. Soil pH (water, 1:2 ratio on fresh soil) and gravimetric water, together with water table depth (dipwells) are measured monthly. Trace gas measurements are made in these chambers (free air volume <8 l) and CO_2 is measured monthly with a PP systems soil respiration chamber that attaches to permanent collars at soil level, with the vegetation and litter scraped back. In addition a number of soil enzyme assays linked to litter decomposition and nutrient mobilization have been made through collaborative student projects.

4.1 Impacts

4.1.1 Species response to ammonia and consequences including soil changes

Over the first five years of this N manipulation the most significant visible changes have occurred in response to ammonia and in the wet N plots receiving 56 kg N $ha^{-1}a^1$ as oxidised N with P+K. As the years of NH_3 exposure have increased so has the distance from the NH_3 source over which the sensitive species *Cladonia portentosa, Calluna*, and *S. capillifolium* have been irreversibly damaged, showing that repeated exposure to NH_3 causes these plants

to succumb to ever lower NH_3 concentrations. Our results suggest the damage is concentration driven, actual exposure concentrations being much higher than indicated from the monthly monitored value, which represents an integrated concentration based on the whole exposure period, not just when the NH_3 was being released. The mechanisms of damage appear to be species specific, ranging from direct toxicity in *C. portentosa* to a combination effect for *Calluna* and *S. capillifolium*, apparently linked to drought stress implying NH_3 exposure negatively affects water use efficiency, increasing susceptibility to desiccation (Sheppard et al submitted). As the length of exposure to NH_3 has increased, separating the effects of concentration from accumulating N dose becomes less meaningful. Under this experimental simulation it has been possible to attribute damage symptoms such as bleaching to cause, NH_3 exposure. Annual percent cover monitoring has confirmed that the lichen and cover of green *Calluna* shoots have declined significantly within 30 m of the source over the last 5 years. However, not all ericoid species have been affected negatively, *Erica tetralix*, *Vaccinium myrtillus* and *Empetrum nigrum* are thriving, as is *Sphagnum papillosum* and *S. fallax* which inhabits a wetter niche than *S. capillifolium*. However, at this stage we cannot confirm that the increase in these species is due to the loss of these other species.

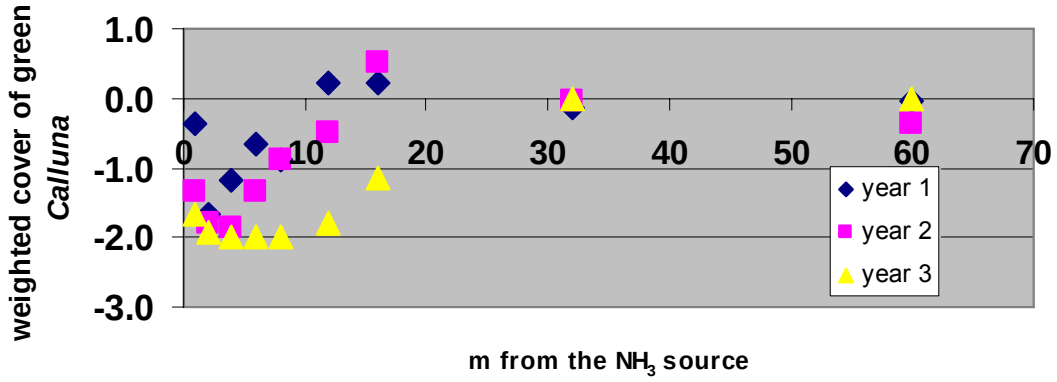

Figure 3. Loss of cover of green *Calluna* shoots along the ammonia release transect, a value of zero signifies no change and negative values indicate a reduction in cover.

In the area of high [NH_3] > 30 µg m^{-3}, up to 20 m from the NH_3 source, there are only remnants of brittle, bleached *Calluna* stems with the occasional green shoot, most of the *Sphagnum* and *Cladonia portentosa* is indistinguishable. The canopy is very open and the hummocks are starting to disintegrate but the *Eriophorum vaginatum*, and *Erica tetralix* remain, although not obviously increasing their cover. However, *Vaccinium myrtillus* and *Empetrum nigrum* are doing very well. New seedlings of *Calluna* and the occasional tiny clump of *Cladonia portentosa* are reappearing as the layer of litter reduces, likewise the *E. angustifolium* appears to be taking advantage of the openness and baring peat. In this region the increases in peat pH and available (mineral) N have also been greatest. The increase in available N including nitrate –N suggests the higher pH resulting from NH_3-N deposition (alkaline gas) has improved conditions for nitrification. Early results also show that trace gas, N_2O release was enhanced while CH_4 emissions were reduced. Establishing cause and effect will be difficult due to the large number of factors that have been influenced by enhanced

NH_3 concentrations; death of *Sphagnum*, recognized for its acidifying properties (Van Breemen 1995) and N immobilisation (Limpens 2003) may be a crucial driver here.

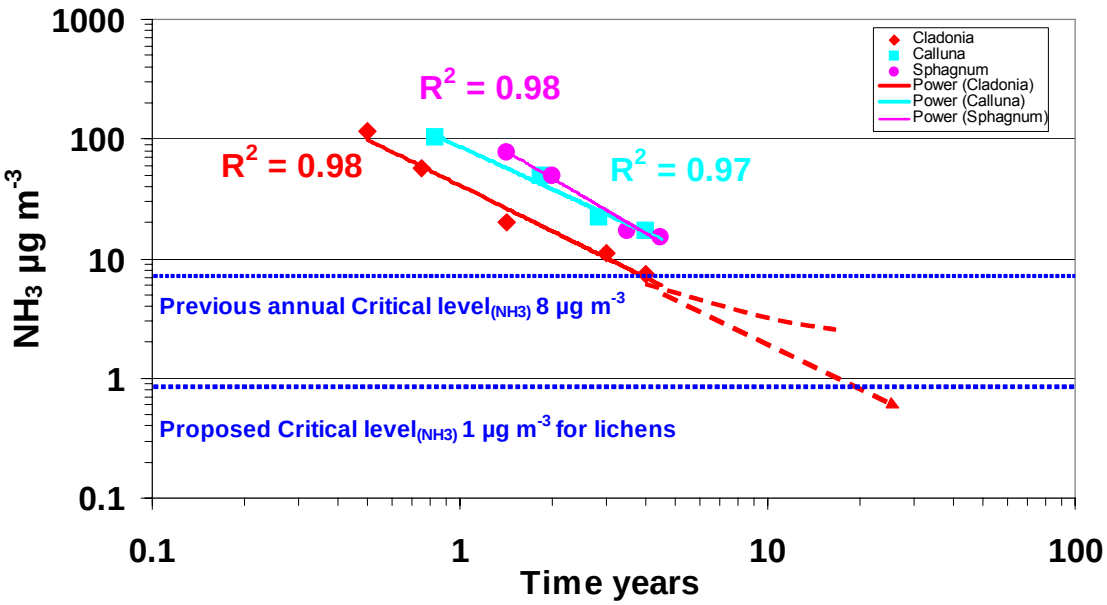

Figure 4. Ammonia concentration causing the death of at least 85% of *Cladonia portentosa, Sphagnum capillifolium* and *Calluna* decreases as the duration of exposure increases.

4.1.2 Responses to wet deposition: species effects and N status

While the effects of wet deposition have been less dramatic, there are signs that the ecosystem is unable to buffer the additional N in the long-term. After 3 years the increase in amino acids in *Sphagnum* was indicative of negative effects, although a change in cover or death were not recorded. This year we are measuring growth and viability in the wet treatments, as cover may not change immediately even though the clump is dying and ceasing to sequester nutrients. Nutrient concentrations, measured in October 2006, in the recent green tissue of *Calluna, Sphagnum* and *Hypnum* have revealed species specific responses that were N form dependent. *Calluna* shoot extension was still responding positively to N dose, but N form had no effect. N status was enhanced but compared with ammonia the effect was >3 fold smaller and insensitive to N form. This was unexpected as shoot uptake of reduced N is almost double that of oxidised N, and shoot uptake of NH_4^+ can account for most of the N applied to the foliage (Bobbink and Heil 1993). However, given the difference in base cations was also small, maximum 15% and the change in N status = 0.03% per 10 kg N added, a form effect would be difficult to detect against natural variation.

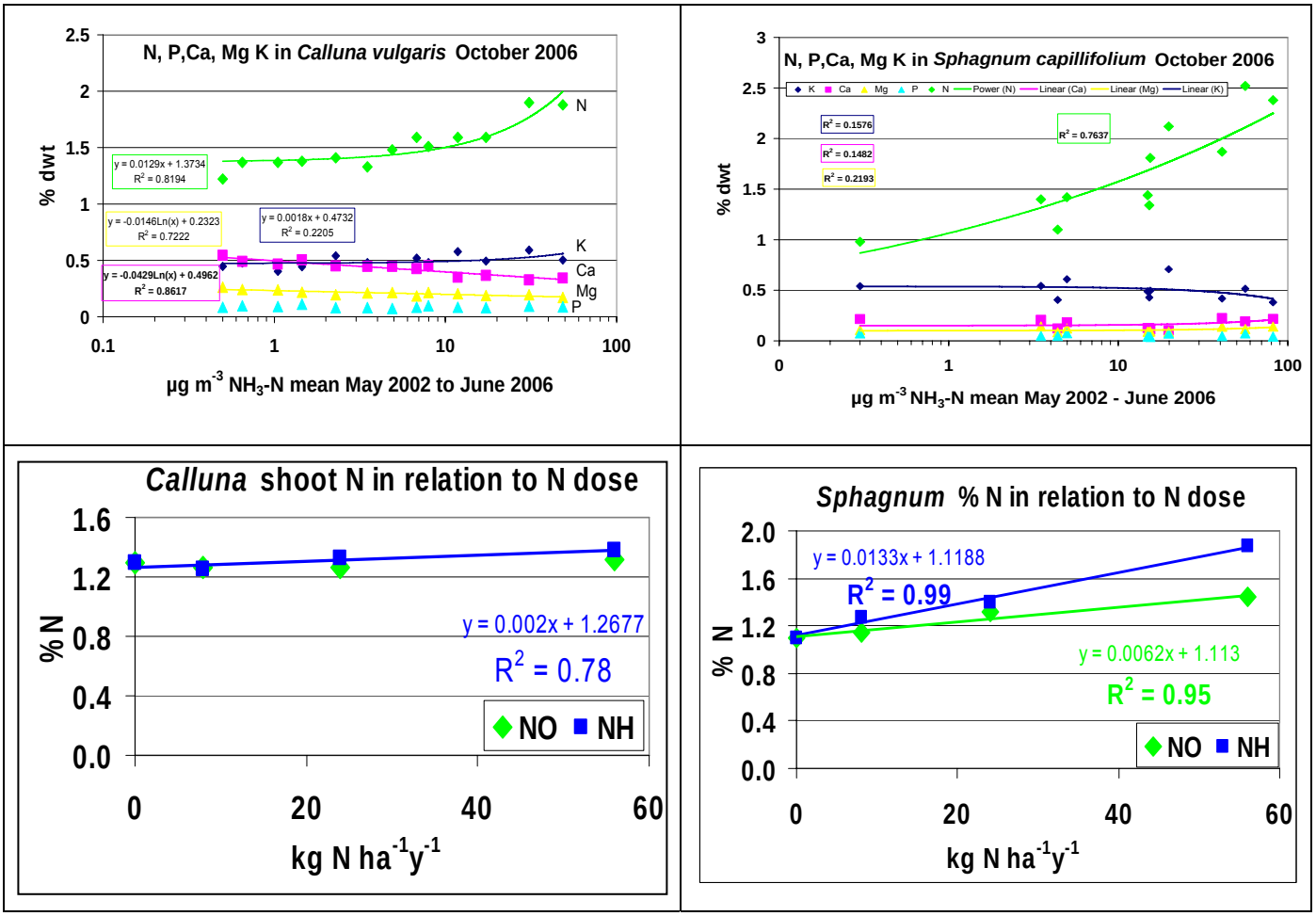

Figure 5. Changes in N, P, K, Ca and Mg concentrations in *Calluna* and *Sphagnum capillifolium,* sampled after 4.5 years exposure to a gradient of ammonia concentrations. N concentrations in relation to wet N deposition, as ammonium or nitrate, in *Calluna* and *S. capillifolium.*

By comparison the nutrient status of *S. capillifolium* was very sensitive to enhanced N deposition and with NH_3 there appeared to be a memory effect, in 2005 N concentrations increased noticeably when the [NH_3] exceeded 5 µg m^{-3} whereas in 2006 the threshold [NH_3] was >3µg m^{-3}. In response to NH_3 exposure N concentrations were increased far more per kg N deposition than with the equivalent wet deposition, reflecting the significant sink strength of wet acidic *Sphagnum* for this soluble alkaline gas. Wet reduced N increased capitulum N by twice as much as oxidised N and the increases were linear with N dose (R^2= 0.99 and 0.95 respectively). Whether these increases reflect greater uptake of one wet form or the other cannot be clarified until effects on growth are established, it is possible that the lower N status with oxidised N reflects growth dilution. These results contradict those reported by Lamers et al. (2000) who describe the relationship between capitulum N and N deposition as logistic, with loss of N sequestration above 20 kg N $ha^{-1}a^{-1}$. At Whim *S. capillifolium* appears to tolerate significantly higher N deposition/N accumulation without loss of function, however, this may change, given the memory effect. Dry N deposition increased K concentrations R^2 =0.8 in 2005 but not 2006. These increases are difficult to explain unless they represent a negative effect on growth, which would be predicted from the amino acid

concentrations which were significantly enhanced (d'Hooghe 2007). Wet deposited reduced N had much greater detrimental effects on base cation status than oxidised N or ammonia gas, neither of which significantly affected base cation status. Competition for exchange sites (Bobbink and Heil 1993) and increased acidity may explain why the cation NH_4^+ has this negative effect on base cation, particularly Mg^{2+} status. These N driven responses and the differential effects of N form and N dose were exaggerated (greater slopes $R^2 = 0.95$ and 0.99 respectively) in *Hypnum* moss except that the difference in N status between reduced and oxidized N was smaller.

4.1.3 Responses of soil chemistry

Effects on soil chemistry on this acid peat have indicated a high degree of between plot variability, so until all the monthly samples have been analysed, observations remain preliminary, with the exception of pH. pH trends are relatively stable, with nitrate additions and the high [NH_3] increasing pH and ammonium additions reducing pH. Soil water N concentrations have responded to the different N treatments and 0.02-0.14 mg N l^{-1}concentrations of mineral N have been measured in soil water, except where the N additions include P+K, which results in lower N concentrations. Dissolved organic carbon was increased where [NH_3] were high and there appears to be a reduction in DOC with high wet reduced N and an increase with high wet oxidised N. These data will be reported fully under the next Umbrella, when monthly sampling has been concluded.

5. Discussion and implications

Ombrotrophic bogs and peat are synonymous with the vegetation they support, and have high conservation value as well as representing a major global carbon sink. The UK has a relatively large proportion of this important resource and its state and ability to sustain the carbon sink function is giving cause for concern (Thompson et al. 1995). A role for enhanced N deposition in the loss of function could be expected since the keystone peat formers are Sphagnaceae and Cyperacae that have evolved with a low nutrient requirement, and in the case of *Eriophorum*, reliance on organic N. Results from Whim show that intensive livestock units pose a real and significant threat to the keystone plant species on which the existence of this valuable resource depends. So far we have insufficient data to comment on the effect of ammonia on the functioning of peat as a carbon sink or as an emitter of greenhouse gases. On this acid peat bog the loss of species associated with elevated ammonia concentrations has not resulted in invasion by graminoids or other higher plants that might affect the water table balance, which appears to be a key factor controlling greenhouse gas emissions on this site. The significance of wet deposition for the sustainability of peat based ecosystems such as ombrotrophic blanket bogs remains unclear at this time. The increase in amino acid concentrations in the S*phagnum*, together with the detection of mineral N in the soil water, suggests the ability of *Sphagnum* to sequester the mineral N is finite and may have been exceeded (Lamers et al. 2000). Over the next few years we should know the extent of the threat posed by wet N and whether N form is important.

5.1 Outputs and collaboration

The most significant output from the Whim experiment has been the central role it played in the revision of the critical Levels for ammonia: The new critical levels of ammonia have now been adopted by i) the UNECE expert workshop, ii) the ICP Vegetation, ii) the ICP

Modelling & Mapping, with the new text to be inserted soon into "the Mapping Manual" - which is the standard reference, iv) the Working Group on Strategies and Review noted the outcome of the workshop without comment i.e. accepted. See ECE/EB.AIR/WG.5/86: "30. (g) Noted the conclusions and recommendations of the Workshop on Atmospheric Ammonia held in December 2006 in Edinburgh and invited the EMEP Steering Body and the Working Group on Effects to consider these at their sessions in August and September 2007" see: http://www.unece.org/env/documents/2007/eb/wg5/ece.eb.air.wg.5.86.pdf

This work was presented at the ESF meeting on reduced N in Austria (October 2007), and in formal seminars at Lisbon University, Portugal (March 2006) by invitation. The site has played host to many visitors from SNH, *e.g.* area officers involved in advising SEPA on IPPC implementation and SEPA re monitoring for IPCC licencing and overseas. It is used by researchers and students from the Universities of Bangor, York, Aberdeen, Manchester Metropolitan, Nottingham, Lisbon, Nijmegen, Macaulay Institute and SAC.

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6. Possible future work

1. Maintain NH_3 fumigation and monitoring; Five years on the detrimental effects of ammonia exposure continue to accumulate, leading to lower NH_3 concentration thresholds, we need to know if concentration thresholds will continue to fall, by how much and if other species will start showing damage.

2. Maintain the wet deposition; An N deposition experiment on an ombrotrophic bog in Canada, with similar ambient N inputs (Bubier et al. 2007) found that after 5 years wet deposited NH_4NO_3 at 60 kg N $ha^{-1}a^{1}$ significantly reduced *Sphagnum* cover. We are at that critical time, and despite significant changes in *Sphagnum* biochemistry have not recorded large reductions in *Sphagnum* cover. Potential interactions with climate, via growth mediation and experimental regime may be responsible but in order to validate the use of empirical evidence in the setting of Critical Loads for N we need to explain why these different responses occur.

3. Determine if and how the type of exposure to wet and dry deposition influences the effect. Earlier open-top chamber studies on conifers showed that ion concentration and exposure frequency also significantly influenced the response to wet deposition (Sheppard et al. 1993). Given that the ion concentrations and exposure frequencies in most manipulation experiments are significantly higher/lower respectively than either rain or cloud water, we need to establish if and by how much responses are over or under estimated. Likewise, for ammonia exposure we need to evaluate the significance of exposure pattern, as most of the studies evaluating the species sensitivities have generated concentration thresholds based on continuous exposure at constant concentrations, whereas field exposures are normally intermittent, consisting of short periods of high concentrations, often followed by extended periods of ambient concentrations.

4. Establish why some species of moss tolerate NH_3 and accumulate high N concentrations whereas others are highly sensitive. Moss N status is potentially a robust N indicator, even discriminating N form, but we need to investigate the suitability of different species and why some are N accumulators if we are to make widespread use of them for regulatory purposes.

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Appendix List of responses measured with arrows to indicate the direction of significant changes, with ns signifying non significant change, and empty boxes = not assessed/obvious.

WET-N dep^{n}	2004	2005	2006
Calluna			
shoot extension	Form NH↑	Dose ↑	Dose p<0.05, Form NH↑, p=0.095
shoot cover,	Dose ↑ ~20%		
Visible damage	none	none	
Frost Tolerance	Feb ns		
Foliar N	Feb ns		Oct Dose 8↓ 56↑
C/N Ratio			ns
Arginine		ns	NH56>NO56
K			NH↑
Mg, Ca			ns (trend ↓)
Sphagnum			
Visible damage	none	none	Dose↑ NH>NO
Capitulum N	Dose↑ NH>NO	Dose↑ NH>NO	Dose↑ NH>NO
K			Dose & NH↓
Mg, Ca			↑ dose NO ↓ dose NH
Cover	ns		
Arginine, asparagine, glutamine		Dose↑ NH>NO	
Hypnum			
Visible damage	none	none	Dose↑ NH>NO
N	Dose↑ NH>NO	Dose↑ NH>NO	Dose↑ NH>NO
K			Dose↓
Mg, Ca			↑ dose NO ↓ dose NH
Cladonia			
Visible damage		none	Dose↑ NH>NO
% N		Dose↑ NH>NO	
Soil			
Extractable NH_4^+		↑	ongoing
Extractable NO_3^-		↑	ongoing
'Rhizon' NH_4^+		↑	ns
'Rhizon' NO_3^-		↑	ns
Mineralisable N		↑ dose	ongoing
Gaseous N_2O			ongoing

Gaseous CH_4			**ns ↓ N dose**
AMMONIA	**2004**	**2005**	**2006**
Species loss, NH_3 μgm^{-3}			
Cladonia	**20**	**10**	**7**
Calluna	**45**	**25**	**15**
Sphagnum	**38**	**20**	**12**
(Hypnum)	**38**	**38**	**60**
Recolonisation			**Cladonia & Calluna**
Species expansion	**Erica tetralix, Empetrum nigrum, V. myrtillus**	**E. tetralix, Eriphorum angustifolium, E. nigrum, V. myrtillus**	**Erica tetralix, E.angustifolium, Racomitrium lanuginosum, E. nigrum, V. myrtillus**
Calluna			
Green shoot cover,	**↓ > 45 μgm^{-3}**	**↓ -> 25 μgm^{-3}**	**↓ > 15 μgm^{-3}**
Visible damage	**Yes pathogen**	**Yes pathogen**	**Spring frost**
Frost Tolerance	**↓> 80% explained by NH_3**		
Foliar N		↑	
C/N Ratio		↑	
Sphagnum			
% N	↑	↑	↑
Chlorophyll		↑	
Arginine		↑	↑
K			↓
Mg, Ca			↓

Work Package 2:
Impacts, Recovery and Processes

Task 8:
The potential for recovery and interactive effects of climate change in acid grasslands and heathlands

A. Sowerby, B. Emmett, M.James
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Centre for Ecology and Hydrology Bangor

Task 8 - The potential for recovery and interactive effects of climate change in acid grasslands and heathlands

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1. Summary

This project aims to evaluate the rate and direction of recovery of a polluted upland ecosystem and identify potential implications of future climate change. An experimental approach was used that used retractable curtains (Beier et al. 2004) to reduce N deposition, create repeated summer drought and produce whole ecosystem warming. The first site was located on a heavily polluted heathland in the Peak District. Here repeated summer drought, warming and a pollution recovery treatment were established; unfortunately, the treatments were not in place for sufficient time to gain any meaningful results within this reporting period. A second field site with a record of 8 years of climate change manipulation on a heathland in less polluted conditions has continued. Results from this site have highlighted the importance of repeated summer drought in mobilizing both carbon and nitrogen from the soil store as well as indicated some interesting indirect affects of the climate treatments. Results from this site continue to be presented at a range of national and international conferences and meetings and future work will integrate the results to inform both model development and critical load assessments. The implications of the potential impact of climate change for critical loads and predicting ecosystem change are given.

2. Policy Relevance

Substantial areas of the UK are currently identified as being in exceedance of empirical critical loads of nutrient nitrogen. Emission control policies are intended to decrease deposition loadings and thus initiate recovery from N enrichment. Climate change may interfere with this recovery and/or mask signals due to both direct and indirect effects on N cycling and the competitive balance between species. Whilst monitoring work is undertaken to follow ecosystem responses due to changes in air pollution and climate change, concurrent changes in drivers such as land management, air pollution and climate can make attribution of change in ecosystem structure and function to individual drivers problematic. Experimental approaches such as this undertaken here enable these causal links to be isolated and quantified. This type of information is required by both model developers and the UK National Focal Centre on Critical Load Mapping to enable the potential implications of deposition reductions and climate change on critical loads exceedance to be evaluated

3. Objectives

- Evaluation of the recovery of damaged ecosystems in response to N deposition reductions and interactive effects of climate change
- Assessment of the processes controlling nitrate leaching from soils to freshwaters and rates of leaching

- Effective collaboration with researchers under the Freshwater and Dynamic modeling Umbrellas

4. Methods and results

4.1 Study sites

The Climoor fieldsite in NE Wales has monitored the long term (8 years) affect of experimental summer drought and warming using an automated retractable roof system. The community is a typical north-western Atlantic upland heathland dominated by *Calluna vulgaris*. Summer drought is created using transparent plastic roofs that cover the experimental plots whenever rain is detected. The treatment operates June-September annually and reduces soil moisture levels to approximately 10 m^3 m^3, excluding on average 20% of annual rainfall. The warming treatment uses a reflective curtain which covers the plots at night, trapping heat absorbed by the community during the day. On average, the treatment results in a 0.5 ^oC increase in soil temperature and a 0.8 ^oC increase in mean daily air temperature. Beier et al. (2004) gives more details on the experimental methodology. The Climoor experiment has been further developed in a new field site in the Peak District (the Peaknaze Recovery roof site); where as well as climate manipulations, an irrigation system used in conjunction with the automated retractable roofs provides a pollution recovery treatment. In the pollution recovery treatment rainfall is intercepted from the plots, and "clean rain" (a chemically created solution with nitrogen and sulphur not included) sprayed simultaneously over the experimental plots. Unfortunately, we experienced a number of difficulties at the Peaknaze site during its relocation from mid-Wales and rebuild in the Peak District. This has meant that treatments were not applied in sufficient time for any meaningful results (derived from the application of the treatment) to be gained within this reporting period. The Peaknaze field site has higher levels of wet N deposition (on average 16.4 Kg N ha^{-1} yr^{-1}), compared to 10.4 Kg N ha^{-1} yr^{-1} at the Climoor site.

Although heathlands are characterized by low nutrient availability, large pools of N are stored in the soil profile. The characteristic low temperatures and high rainfall of upland heathlands limit many aspects of biological activity. With climate change predicted to lessen these key limiting abiotic factors, there is potential for climate change to result in an increase in many biological processes, such as N mineralisation. Aspects of climate change, such as warming, may therefore be important to consider when predicting recovery from enhanced N deposition. If available nutrients increase within a habitat such as heathland, which is characterised by its ability to exist in conditions of low nutrient availability (be it from increased N availability from N deposition or from a climate related impact on biological processes) a number of important affects are key. Changes in the plant community as the competitive drivers between species change is one possible outcome; in heathlands the replacement of *Calluna vulgaris* with the grass *Deschampsia flexuosa* has been predicted with increased N availability, particularly during regeneration (see task 4 in work package 2). Increased foliar N leading to increased susceptibility to secondary stressors (such as herbivory or pathogen infection) has also been predicted as an impact of increased N availability (Bobbink et al, 1998). Figure 1 shows a simplified schematic of the nitrogen and carbon cycle, our aim has been to follow a whole ecosystem approach to observe the impact of climate and N input (pollution recovery) treatments.

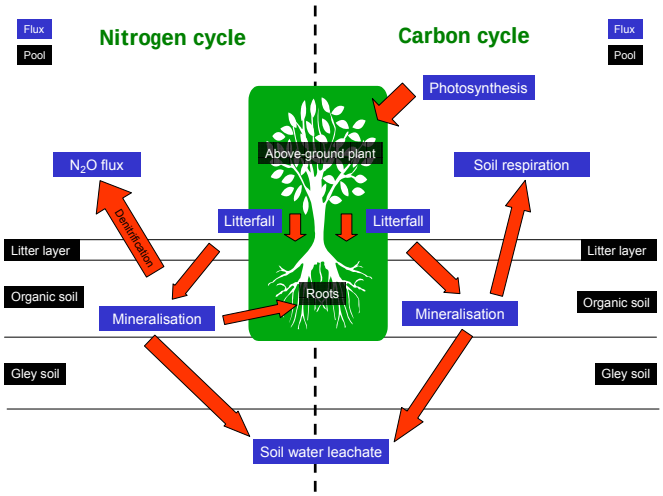

Figure 1. Simplified schematic of the nitrogen and carbon cycle.

4.2 Experimental results

4.2.1 Soil net N mineralization

Data from N mineralisation studies (comparing NO_3 and NH_4 concentrations in soil cores, before and after an incubation period in situ) at the Climoor site have produced results with a high degree of variability, therefore we have been unable to pick up an increase in net N mineralisation from the warming or drought treatment. We believe this is a methodological problem however, as clear moisture and temperature responses (and thresholds) have been observed for mineralisation in heathland soils (Emmett et al, 2004). Figure 2 shows results from an inter-site comparison of net N mineralisation of heathlands across Europe.

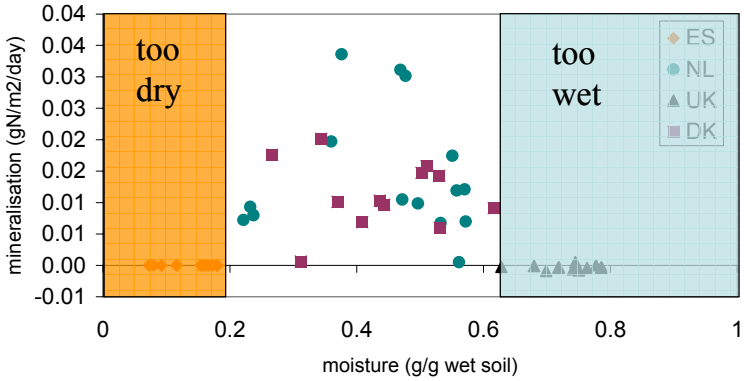

Figure 2. Net N mineralisation and soil moisture content in European heathland soils. Redrawn from Emmett et al, 2004.

4.2.2 Soil water chemistry

At both the Climoor and Peaknaze site, soil water nitrate concentration from the organic soil horizon has been measured over a number of years through the use of zero tension tray lysimeters. Climate manipulation affects have found to be minimal, and differences between the two sites out weigh any differences resulting from the drought or warming treatment at Climoor (Figure 3). Soil water concentration of NH_4, pH and acidity were also all found to not be influenced by the summer drought or warming treatment at the Climoor site.

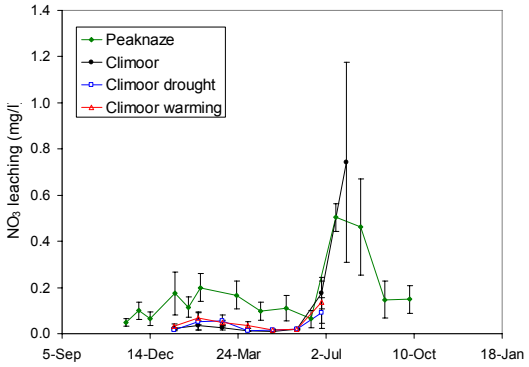

Figure 3. Mean soil water nitrate concentration from the organic soil horizon at the Climoor and Peaknaze sites in 2006. Error bars show the standard error of the mean, n=3.

4.2.3 Foliar N concentration

As with the measurements of soil water nitrate, foliar N concentrations were also unaffected by the climate manipulation treatments (Figure 4).

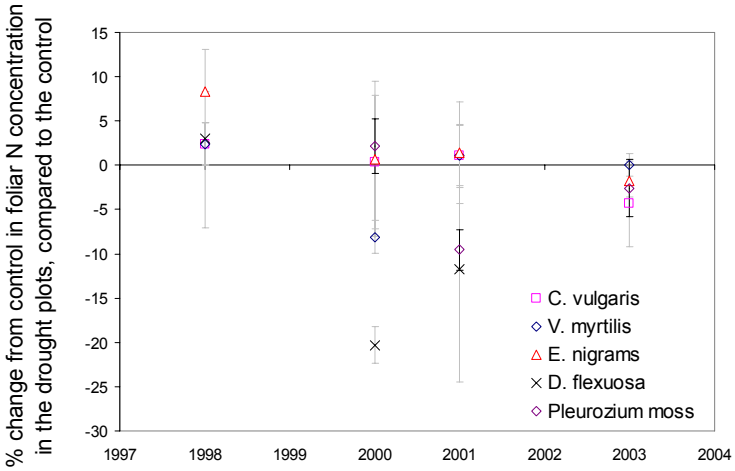

Figure 4. Foliar N concentration in the different plant species at the Climoor field site, expressed as percent change from control in the drought plots. Error bars show the standard error of the mean, n=3.

4.2.4 Plant growth responses

The lack of difference in foliar N was despite differences in growth responses to the climate treatments (Figure 5), the growth responses have continued beyond 2003, for example, the warming treatment continued to result in a significant reduction shoot length in 2006 (and presumably therefore vegetation biomass) of *Empetrum nigrum* (Figure 6, two-way ANOVA, F=10.54, p=0.014).

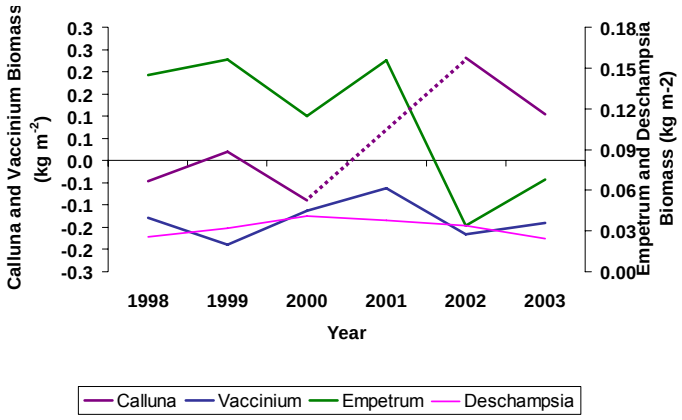

Figure 5. Impact of the drought treatment on vegetation biomass, expressed as percent change in biomass in the treatment plots, compared to the control, n=3.

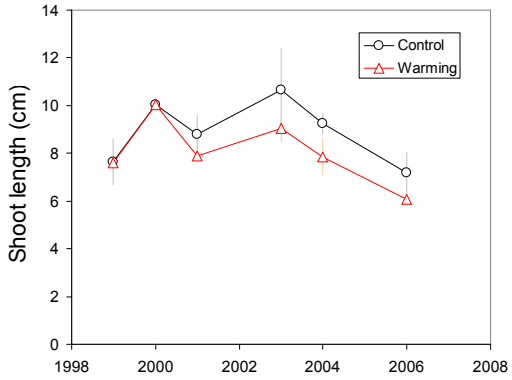

Figure 6. Annual growth rate of *Empetrum nigrum* (as described by the length of the current years shoots measured in mid-August approximately annually). Data includes *E. nigrum* growing in both the control and warming plots at the Climoor site. Error bars show the standard error of the mean, n=3.

4.2.5 Plant physiology and fungal pathogen

One of the more unexpected results from the Climoor field site was that over the number of years that the drought treatment had been in place, and all the changes that were observed as a result of it, one species failed to show the predicted response in vegetative biomass increase, based on physiological measurements. When measured in 2005, *Vaccinium*

myrtillus displayed significantly greater rates of photosynthesis (paired t-test, p=0.001) and water use efficiency (paired t-test, p=0.026) in the drought plots (albeit outside the drought period) suggesting more favorable conditions for growth with the drought treatment. In additional, when measurements of photosynthesis in *V. myrtillus* were considered over a range of soil moistures at the site, rates were found to increase with reduced soil moisture (Figure 7). This further suggested that the soil moisture levels at the site were greater than the optimum for *V. myrtillus* growth and in theory the drought treatment should improve conditions for growth. However, when biomass of *V. myrtillus* was measured in the plots, using pin-point methodology, no change in *V. myrtillus* biomass was observed (Figure 5, shoot length data for *V. myrtillus* not shown).

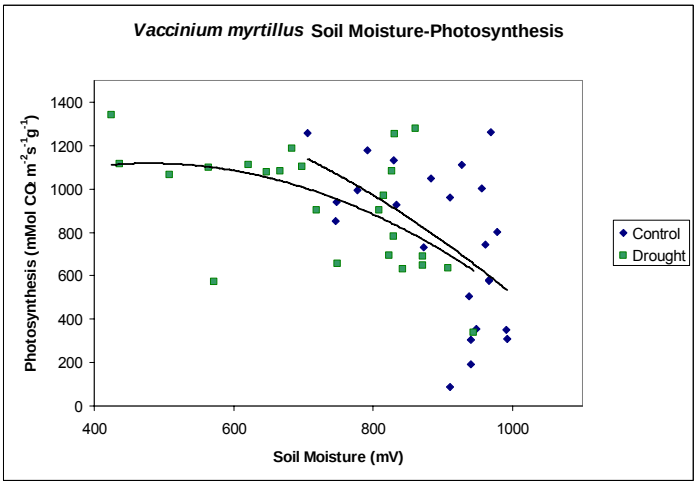

Figure 7. The relationship between rates of photosynthesis in *Vaccinium myrtillus* and soil moisture at the Climoor site, n=3.

When plant pathogens were considered, the drought was found to increase the severity of the *V. myrtillus* fungal pathogen *Pucciniastrum vaccinii*. (Figure 8, Chi-squared test, p=0.004) thus explaining the lack of response in *V. myrtillus* biomass. This also highlights the importance of studying the impact of climate or pollution drivers on intact ecosystems; it is unlikely we would have observed this important interaction from a pot-based lab/greenhouse experiment.

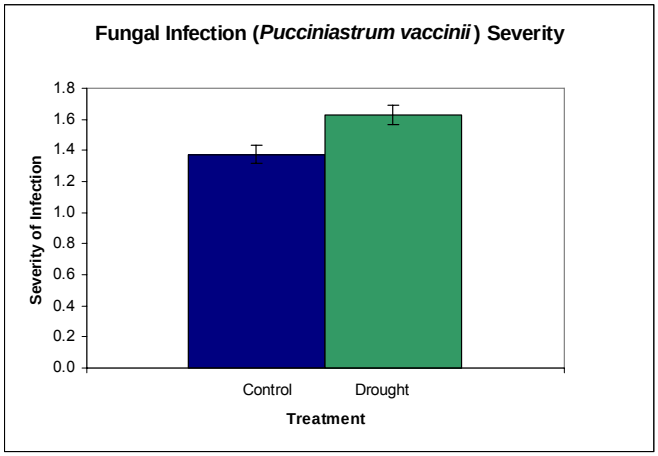

Figure 8. Severity of fungal infection (*Pucciniastrum vaccinii*) in *V. myrtillus* in 2005. Significance reported on a fungal infection severity scale of 0 = no evidence of infection, 1 = mild, 2 = moderate, 3 = severe. Error bars show the standard error of the mean, n=3.

4.2.6 Changes to the Carbon cycle

Concentrations of DOC in the soil water in the organic horizon in Climoor and Peaknaze were remarkably similar; however unlike nitrate and the majority of other elements considered the drought treatment increased concentrations of DOC in soil water (Figure 7).

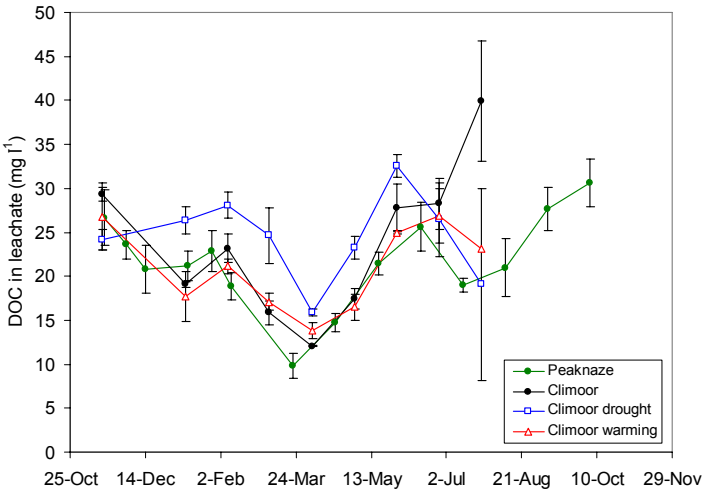

Figure 9. Mean soil water DOC concentration in the organic horizon at the Climoor and Peaknaze sites in 2006. Error bars show that standard error of the mean, n=3.

At both sites, peaks of DOC in the organic horizon soil water were observed from across all plots during the late summer, early autumn with lowest fluxes observed in late winter, early spring. The drought treatment at Climoor significantly increased the maximum value of these peaks (Figure 8, Paired t-test, p=0.006)

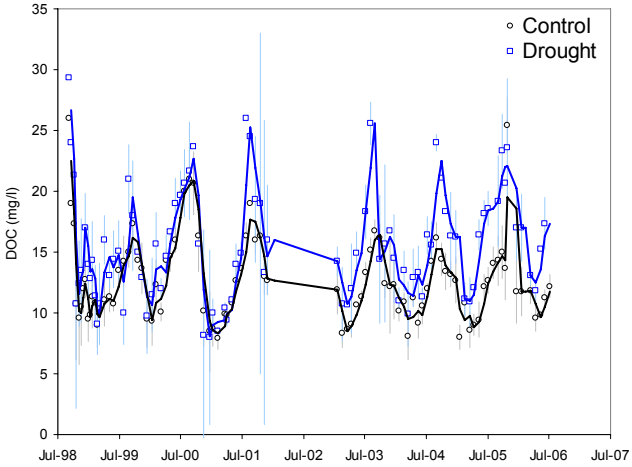

Figure 10. Mean DOC concentration in the soil water in the lower soil profile (collected in suction samplers at a 30cm depth) over the 8 years the treatments have been applied at the Climoor site. Error bars show the standard error of the mean, n=3.

Along with increased DOC concentrations in soil water, soil respiration was also found to be significantly increased by the drought treatment (Figure 9, Genstat 8., repeated measures ANOVA, p=0.034), indicating large changes in the C cycling in the drought plots at the site. This was also observed from the warming treatment. The warming treatment also resulted in a 30 % reduction in the growth rate of the shrub *E. nigrum*. The warming treatment produced a 0.8 ^{o}C increase in air temperature, well within predictions for the next 30 years, and yet this increase in temperature could result in a change in the plant community composition at the Climoor field site. Of particular note is a lack of acclimation or recovery in many of the changes seen as a result of the climate change treatments. In particular, a "memory" effect in soil moisture has been observed, where despite over 1000 mm of rain falling outside the drought period, the soils did not rewet to that seen in the control. The year round decrease in soil moisture has impacted on both above- and below-ground C cycling (Sowerby et al, in review).

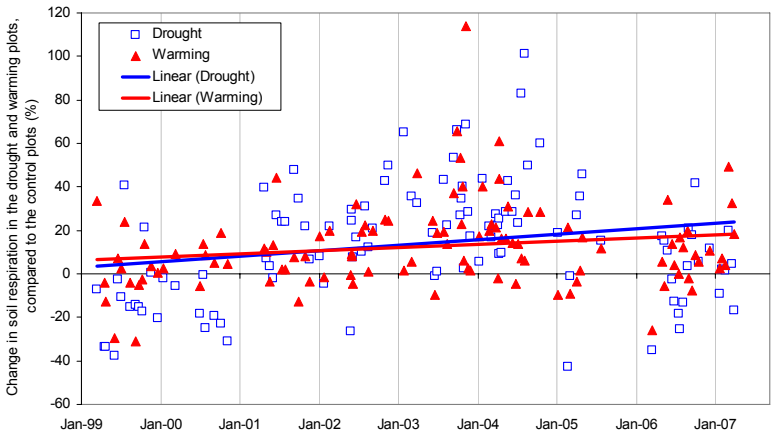

Figure 11. Change in soil respiration in the drought and warming plots, compared to the control plots at the Climoor site, n=3.

5. Discussion and implications

5.1 Discussion

Low availability of nutrients is a key characteristic of heathlands as well as the evergreen shrubs, such as *Calluna vulgaris*, that dominate the plant community. Increased nutrient availability, in particular N, can lead to increased biomass production and N cycling and can result in a replacement of the shrubs with grass species such as *Deschampsia flexuosa*. In forest systems, N input exceeding the critical load has lead to N saturation and resulted in increase in N leaching (Aber et al, 1989). Climatic warming has also been shown to impact on N cycling and N leaching (Schmidt et al, 2004). Although heathlands are characterized by low nutrient availability, large pools of N are bound up the soil profile and could be released with altered climatic conditions. For example, Wright (1998) observed a 40 % increase in nitrate leaching as a result of a warming and elevated CO_2 manipulation in an N-saturated boreal forest.

The climate manipulations (warming and summer drought) at the Climoor field site did not result in increased N leaching. Plant biomass increased in response to both the warming and

drought treatment without an observed change in foliar N concentration. This suggests that the community is not N saturated and any excess available N released as a result of the climate treatments was taken up by the vegetation at the site, rather than leached out of the system in soil water.

Vaccinium myrtillus, the bilberry plant, failed to show an increase in biomass in response to the climate treatments, despite observed beneficial changes in its physiology and increased N supply. The lack of response in biomass can be linked to an increased incidence of the fungal plant pathogen *Pucciniastrum vaccinii*. In a similar fungal pathogen of Vaccinium, *Valdensia heterdoxa*, Stengbom et al (2002) have linked infection to higher concentrations of free amino-acids in the Vaccinium leaves. Thus it is possible that the increase in the degree of infection is linked to increased N availability. Further work at the site aims to corroborate this. In a sister site in Denmark, with an identical experimental set-up to the Climoor site, the warming treatment resulted in significant increase in foliar N. The increase in foliar N has been linked to a heather beetle attack that greatly reduced plant biomass in the plots (Penuelas et al, 2004). In a review of the impacts of N deposition, increased susceptibility to secondary stress factors, such as herbivory or fungal infection, was noted as a key affect of N deposition (Bobbink et al, 1998). Intact ecosystem studies, such as Climoor and Peaknaze, are key in understanding the impact of multiple drivers on ecosystem processes. Many important interactions simply cannot be replicated, and will not be observed, in the lab.

When the Climoor and Peaknaze soil water chemistry data was compared, Peaknaze, a site with a higher rate of N deposition, had greater loss of N through nitrate leaching than Climoor. This does go some way to suggest that as deposition of N and S is reduced, a similar reduction in NO_3 leaching may be observed. This is further corroborated by collating published results on nitrate leaching from UK heathland habitats, a clear relationship can be seen, with a threshold somewhere around 20-25 Kg N ha yr^{-1} above which leaching occurs from heathland soils (Figure 10). This agrees approximately with the conclusions of Dise & Wright (1995) which looked at the link between N deposition and nitrate leaching from European forests.

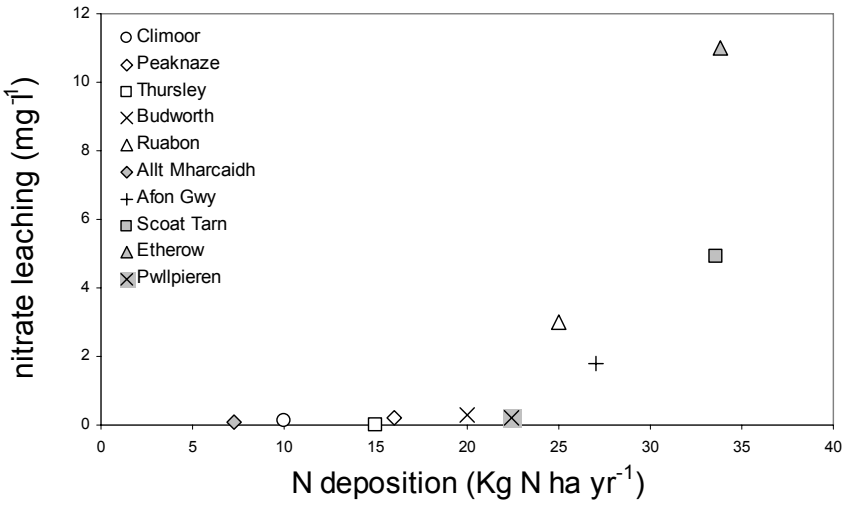

Figure 12. Nitrate leaching from heathland soils. Data from the Climoor and Peaknaze field sites, as well as from Thursley, Budworth and Ruabon (taken from Power et al, 2004) and Allt a' Mharcaidh, Afon Gwy, Scoat Tarn and River Etherow (taken from Curtis et al, 2004).

However, comparing different sites under different N depositions is not the same as assessing recovery. Recovering sites have had prolonged exposure to N enrichment and may have larger pools of N stored within the soil profile and prolonged leaching beyond a reduction in N deposition is perhaps likely. Unfortunately, we have no experimental treatment evidence from the Peaknaze recovery plots to prove or disprove this as yet, the recovery treatment has not been in place long enough to derive any meaningful results. It is worth noting that the difference in nitrate leaching from the Climoor and Peaknaze sites far exceeded any differences resulting from the climate change treatments at the Climoor site, however.

Carbon has been shown to be key factor in nitrate leaching from soils and a relationship between DOC and NO_3 has been demonstrated in surface stream waters (Evans et al, 2006). Despite having little impact on soil nitrate concentrations in the organic horizon of the soil, the drought did impact on the concentration of DOC in the soil water from both soil horizons, as well as increased rates of soil respiration and increased plant growth. The drought treatment resulted in a year round reduction in soil moisture over a number of years at the site, and impacted severely on many key C stores. The warming also increased rates of soil respiration and reduced the biomass of one of the main shrub species at the site *Empetrum nigrum*. If the results from the Climoor site are replicated across all moorlands in Wales and the North of England over the next few decades we may witness the local extinction of this species in its most southerly distribution. Many of the results gained from the Climoor site took years to materialise, it is vital to continue long-term climate manipulation studies.

Although the two indicators of N enrichment, increased N leaching and foliar N, were unaffected by the climate manipulations at the Climoor site, it is important to note that increased N availability and changes in the N cycle were indicated. Plant biomass was stimulated without a concurrent change in foliar N concentration, this means in terms of mass balance of N, more N sequestered into the above-ground plants at the site. Along with increased N uptake by the plants at the site, key aspects of the C cycle at the site indicated change resulting from the climate manipulations. Further work at both sites will work towards directly quantifying the degree of change in the N and C cycle in heathlands as a result of C change and reduced N and S deposition.

5.2 Implications

- The effect of climate manipulations on N and pH concentration in soil water are modest relative to the effect of N input. However, there is evidence of increased N supply to promote vegetative growth.
- The response of one key species was moderated by an interaction with a fungal pathogen. It is unclear whether this was a direct response to climate variable or an indirect response due to increased N supply.
- Despite evidence for increased N supply to plants, no significant change in foliar N or N concentration in soil water suggests these are not early indicators of changes in the N cycle in these heathland systems.
- The climate manipulations mobilized C within the soil profile, in both dissolved and gaseous form. Effect of the repeated drought, in particular, has the potential to significantly reduce soil C stores due to a “memory” effect.

- In terms of the empirically derived critical load, climate x N interactions could significantly increase the risk from pathogens, affecting some but not all plant species occurrence. This could require a decrease in the critical load.
- In terms of the steady state mass balance derived critical load, our results suggest that as climate change progresses, soil immobilisation (N_i) may decline as N availability below-ground was found to increase with warming. Plant uptake/removal (N_{up}) also increased probably in response to the warming treatment and indirectly through the associated increased N supply. N leaching out of the soil (N_{le}) did not appear to be sensitive to climate variables so may remain unchanged. However, Schmidt et al (2004) demonstrated that in a highly polluted Dutch system, NO_3 leaching did increase with warming. Gaseous losses through denitrification (N_{de}) are uncertain, but likely to decline as drier conditions become more common (new measurements under the NitroEurope project will determine this). Overall, the increased supply of N to plants will require a decrease in N critical load.

6. Possible future work

We aim to continue work on the links between the C and N cycle at both the field sites, particularly with the aim of empirically quantifying pools and fluxes of C and N as well as identify and compare changes derived from the climate manipulations and pollution recovery experimental treatments. In the short term, we also plan to further investigate the role of the fungal pathogen and N availability on the growth and physiology of one of the main species at both sites, *Vaccinium myrtillus*.

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Work Package 2:
Impacts, Recovery and Processes

Task 9:
Evaluating the impacts of environmental change using archive vegetation data

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Macaulay Institute

Task 9 - Evaluating the impacts of environmental change using archive vegetation data.

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1. Summary

In this study, archive data collected during the 1960s, 70s and 80s were used as a baseline to allow quantification of changes in montane vegetation and soil composition over the long term. Changes in vegetation composition were correlated with data on N deposition, grazing intensity and climate (temperature and precipitation) change over the same period to assess the relative importance of N pollution, grazing and climate as drivers of change in montane ecosystems. Results showed that montane vegetation has become less variable over the last 30-40 years, with fewer community types represented in the sample. Changes in community diversity were greatest in southern and eastern mountain areas which experience rates of N deposition above the current critical load. Species richness had increased in the majority of samples, but in montane communities with normally low richness this may be a deleterious change. Across the whole range of sampled communities, oxidised N deposition was significantly correlated with change in vegetation composition but was not as important as grazing intensity and climate change. Individual communities however displayed varying driver relationships. Oxidised N deposition, deer grazing and climate were the most significant drivers in montane *Calluna* heath composition, while *Racomitrium* heath changes were related to reduced N inputs and sheep grazing. In all communities changes in both vegetation and soil properties were slow. While N deposition is probably a significant driver of vegetation change in montane communities, management and policy recommendations must take into account the equally significant impacts of land use and climatic changes.

2. Policy Relevance

This work aims to provide evidence for the impact of nitrogen deposition on potentially sensitive montane communities which are of high conservation value but are relatively little researched. It also aims to explore the relative roles of nitrogen deposition, climate and land management in driving observed vegetation changes.

3. Objectives

- Evaluation of the impacts of nitrogen deposition on montane ecosystems and the possible interactions with other variables.

4. Methods and results

4.1 Methodology

The aim of this study was to test the hypothesis that '*species composition of montane plant communities in Scotland has changed during the last 30-40 years as a result of environmental and land management changes*'. The study was based on a re-survey of vegetation data for montane communities held in the Macaulay Institute data archive. These data were originally collected by E.L. Birse and J.S. Robertson during the 1960's, 70's and 80's and comprise records of plant species composition and abundance and in some cases soil properties for 677 locations above the potential tree line in Scotland.

In Year 1 of the study, the archive records of montane vegetation composition were converted to electronic format and the species nomenclature brought up to date. The records were also classified into National Vegetation Classification (NVC) communities to allow analysis of the archive data distribution in terms of communities represented, record age and geographical distribution. Resurvey of all 677 records was not practicable, but four target areas with a high density of archive data (Fig. 1) were identified to allow re-recording of the maximum number of sample points within the time available. The selected areas were: the Cairngorm Mountains, the Isle of Mull, Galloway and the Southern Uplands and the Orkney and Shetland Islands. These areas have contrasting climate, land use history and nitrogen deposition rates, but contain similar community types, allowing comparison of changes in vegetation composition between areas exposed to different combinations of driver intensity.

During 2004-6 205 plots were re-located and re-surveyed using the same methodology originally employed by Birse and Robertson (Birse 1980). The original plots were not permanently marked, but were re-located using the site description notes and grid reference on the archive records. For most records, sites and vegetation types matching the archive information were readily found at the original grid references and the accuracy of re-location was judged to be +/- 50m. Vegetation composition was re-recorded at all sites while soils were also sampled at those locations which had been sampled originally, plus a representative selection of additional plots. Soil sampling was done on a by horizon basis, aiming to match the samples with the horizons originally described. In total 82 surface horizons were sampled and 85 subsurface horizons. Soil pH and C:N ratio was measured in all the samples.

The majority of those sites re-sampled were originally recorded during the 1970's giving information on change in vegetation and soil over an approximately 30 year period. Archive and present day vegetation composition data were analysed using a number of different methods. The present-day NVC community of each plot was described so that the range of community types represented by the plots in each area past and present could be compared. Change in species richness of mosses, lichens, higher plants and all species together was also calculated for each plot. Species level changes were investigated using multivariate analysis. In order to determine the potential importance of nitrogen deposition as a driver of change in montane vegetation composition, changes in species composition were correlated with data on total, oxidised and reduced N deposition at each site (R. Smith, pers com) and also with mean grazing animal (sheep and deer) density from 1969-2004 (sheep data from AgCensus, deer data from Deer Commission for Scotland counts) and the 1961-2004 change in seasonal temperature and precipitation (Barnett *et al* 2006).

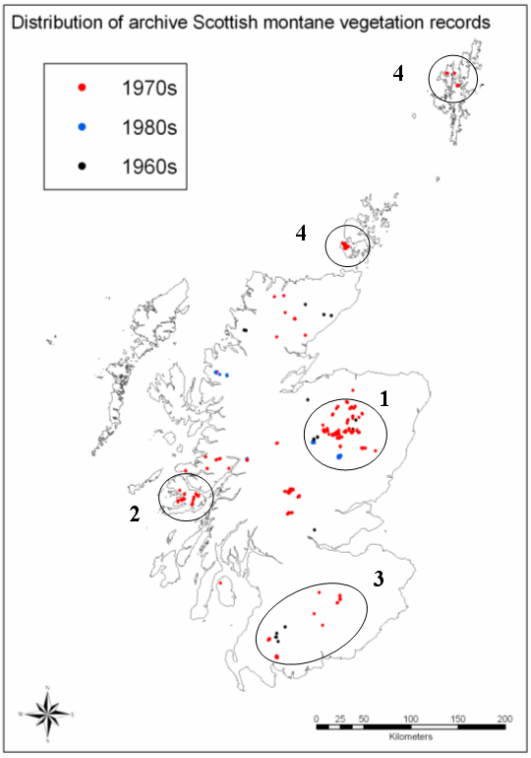

Figure 1. Distribution of archive vegetation records and location of the four target areas for resurvey; 1: Cairngorms, 2: Mull, 3: Galloway and Southern Uplands, 4: Orkney and Shetland Islands.

4.2 Results

4.2.1 Vegetation changes

Analysis of the vegetation data revealed that change in Scottish montane communities over the last 30-40 years is complex, with the nature and magnitude of changes being both community- and site-specific. Some general trends were found however. Changes in vegetation composition had occurred at all of the sites studied, in general these were not dramatic shifts, but more subtle changes in the relative abundance of species. Analysis of the community types (NVC classes) represented by the sample points on the original sampling date and again at the time of resurvey showed that for the Southern Uplands and the Cairngorm Mountains which receive total N deposition inputs of around 10-20kg N $ha^{-1}y^{-1}$, the number of NVC community types present declined from 20 to 17 and from 11 to 8 respectively. On Mull and in the Orkney and Shetland Islands which experience lower deposition rates (of the order of 5-10 kg N ha^{-1} y^{-1}) the number of community types was unchanged (8 and 12 respectively).

Multivariate analysis of the species composition data across all sites and community types showed a similar trend towards reduced vegetation variability (Figure 2). Species composition coalesced over time towards a composition dominated by common, widely distributed species such as *Carex bigelowii*, *Deschampsia flexuosa*, *Racomitrium lanuginosum* and *Vaccinium myrtillus*. Species indicative of specialist habitats such as the

bryophytes *Gymnomitrion conncinatum* and *Racomitrium heterostichum* in snowbeds or the lichens *Alectoria nigricans* and *Flavocetraria nivalis* in exposed heaths appear to have declined. This loss of vegetation variability was not, however, associated with a reduction in species diversity; across all sites and habitats 71% of plots gained species while only 20% of plots showed a decline in species richness.

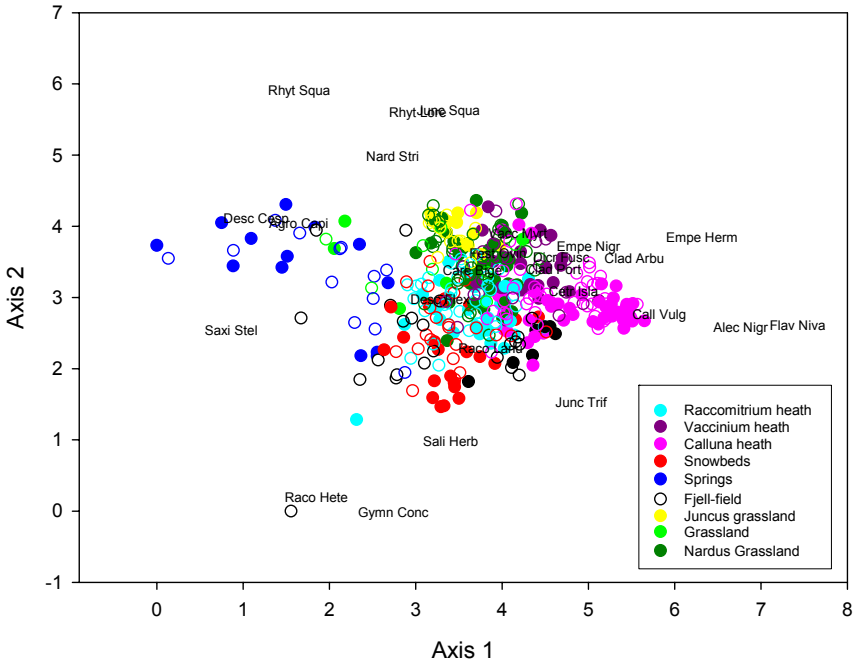

Figure 2. Detrended Correspondance Analysis of archive and new data for all sample points. Colour indicates habitat type, filled symbols are archive data and open symbols show new data. Key species are also shown, indicated by the first four letters of their generic and specific names.

Analysis of changes in individual habitat types highlighted the fact that the direction of vegetation change was highly community specific. *Calluna* dominated montane heaths in the Cairngorms (NVC H13) and in Orkney (NVC H10 & H17) showed opposite trends in species composition suggesting that analysis of change in plant communities should be done on a community-by-community basis rather than by aggregating into broad habitat groups.

Having shown that a measurable change in vegetation composition has occurred in a variety of Scottish montane habitats over the last 30-40 years the next step was to examine the relationships between vegetation and N deposition, climate change and grazing impacts. When rates of change (to account for plot age differences) in vegetation composition on each axis of the ordination were correlated with a range of variables including N deposition (total, oxidized and reduced), grazing animal density and seasonal change in air temperature and precipitation, several significant relationships were found. In the ordination of all community types together (Figure 2), change on the first axis was significantly related to grazer density and summer/autumn temperature change (all $P<0.001$), but no significant correlation with N deposition was found. N deposition was significantly correlated with rates of change on axes 2, 3 and 4 of the ordination. Oxidised N deposition was most significantly correlated with species change ($P=0.014$, <0.001 & <0.001 for axes 2, 3 & 4 respectively); no relationships were found for reduced N deposition.

Links between potential drivers and rates of change were also investigated for *Racomitrium* heath and *Calluna*-dominated heath individually. In *Calluna* heath, oxidised N deposition was significantly positively correlated ($P<0.001$) with axis 1 (interpreted as reduced lichen dominance). Total N deposition and reduced N deposition were also significantly correlated but the relationship was not as strong. Deer density and autumn temperature change (both $P<0.001$) were the other major correlates of change on axis 1 in *Calluna* heath. Change on the other axes in this ordination showed fewer relationships with environmental drivers. Axis 3 changes were also significantly correlated with oxidised N deposition and deer density, while changes on axis 4 were associated with climate. In *Racomitrium* heath change on axis 1 was most significantly associated with sheep density ($P=0.005$), though there was also a significant correlation with reduced N deposition ($P=0.029$) and summer temperature change ($P=0.016$). Change on axis 2 was associated with oxidised N deposition ($P=0.024$) and on axis 4 with reduced N deposition ($P=0.024$). The differences in driver relationships between these two habitats highlight the difficulties in drawing cross-community comparisons and generalizations.

4.2.2 Soil changes

Soils in the montane zone generally lack a true peaty surface horizon, but a shallow H or carbon rich A horizon is usually found at the soil surface. In broad terms the soils sampled were typical of upland Scotland i.e. acid, with surface horizon pH between 4 and 4.5, and with a high organic carbon concentration (generally > 20%). C:N ratio was selected as a potential indicator of long term N impacts on montane soils. Since relatively few re-surveyed sites had archive soil data, robust comparisons were problematic. Rather than making comparisons based on a limited number of matched sample pairs, data from the Macaulay Institute Soil Database was used to make general comparisons through time for soils collected from the same habitat and geographic area. Sufficient archive data existed for this approach to be used for *Calluna-Cladonia* heaths (H13) in the Cairngorms (Table 1) and for *Racomitrium* heath (U10) in the Southern Uplands and Mull (Tables 2 & 3). Archive data were grouped into samples taken between 1960 and late 1970's and late 1970's to mid 1980's and compared with the recent data. Some care must be taken in the interpretation of these data as the older data (prior to late 1970's) were classified as 'montane' only and so may include a wider spread of communities.

Table 1. C:N ratios in near surface soil horizons under *Calluna-Cladonia* heath (H13) in the Cairngorms.

Date	n	minimum	maximum	median	mean
2004	14	20.1	31.1	24.5	24.9
Late 1970s-mid 1980s *(All Scotland for comparison)*	16 *(27)*	16.8 *(9.2)*	59.9 *(59.9)*	33.4 *(26.9)*	36.1 *(28.9)*
1960-late 1970s	31	7.2	76.0	23.5	27.0

Table 2. C:N ratios in near surface soil horizons under *Racomitrium* heath (U10) in the Southern Uplands.

Date	n	minimum	maximum	median	mean
2004	6	14.2	18.2	16.5	16.3
Late 1970s-mid 1980s *(All Scotland for comparison)*	0 *(7)*	- *(9.5)*	- *(22.4)*	- *(16.1)*	- *(15.6)*
1960-late 1970s	16	10.4	20.6	15.8	16.1

Table 3. C:N ratios in near soil surface horizons under *Racomitrium* heath (U10) in Mull.

Date	n	minimum	maximum	median	mean
2004	6	12.5	14.4	13.1	13.3
Late 1970s-mid 1980s	0	-	-	-	-
1960-late 1970s	3	14.4	17.0	15.7	15.7

C:N ratios were higher in *Calluna-Cladonia* heath than in *Racomitrium* heath irrespective of location or date (Tables 1, 2 & 3). Intuitively, this would be expected given the more luxuriant vegetation cover of *Racomitrium* heath compared to *Calluna-Cladonia* heath. There is some suggestion of a decline in soil C:N ratios under *Calluna-Cladonia* heath in the Cairngorms. However, it is surprising that the values are so high in the early 1980s compared to earlier and this aspect requires further investigation. The inclusion of data for this habitat from throughout Scotland (Table 1) brings the values much closer to those for the earlier and later periods. The C:N ratios in Southern Upland *Racomitrium* heath are virtually unchanged between 2005 and the 1960s and 1970s (Table 2). This is somewhat surprising as a decrease might have been expected, given that this area has received a relatively high input of deposited N over this period compared with other montane areas. It must be remembered however that some of the early samples may not be from *Racomitrium* heath.

5. Discussion and implications

5.1 Discussion

5.1.1 Vegetation changes

This study has demonstrated that significant shifts in species composition of montane vegetation have occurred over the last 30-40 years. The principle effect of this has been to reduce vegetation variability, as demonstrated by a reduction in the number of community

types represented by sampling points in southern and eastern mountain areas and a coalescence of community composition towards dominance by common widely distributed upland species at the expense of more specialist species. Such changes are consistent with the predicted effects of N deposition and eutrophication which would be expected to be greater in the southern and eastern sampling areas which experience higher deposition loads (10-20 kg N ha^{-1} y^{-1}) than the north and west (5-10 kg N ha^{-1} y^{-1}). This reduction in diversity has not been seen at the species level, where the majority of sites have experienced increases in species richness. Since many montane communities are relatively species poor, an increase in species diversity does not necessarily represent a desirable change and may be the result of invasion by common and widespread species.

Montane vegetation comprises a series of climax vegetation types, the distribution of which is tightly controlled by the interaction of microclimatic effects and topography. While species change has been induced in N addition experiments on key habitats such as *Racomitrium* heath (Pearce & van der Wal 2002) and *Calluna-Cladonia* heath (Britton & Fisher 2007) the significance of N deposition vs. climate and grazing impacts has not previously been demonstrated at the landscape scale. Correlation of species composition changes with data on change in key environmental variables showed that, across all habitat types, grazing intensity and climate change were associated with the major axis of change in the data while N deposition was associated with lower-order axes. This suggests that at this scale N deposition is not the major driver of change, although it does play a very significant role. It is also interesting to note that oxidised N (which dominates deposition in montane areas away from point sources) was significantly correlated with species change while reduced N was not. Analysis of the species composition data suggested that the direction and magnitude of changes differed between habitats and analysis of links between species change and drivers on an individual habitat basis also showed interesting differences. In *Calluna-Cladonia* heath oxidised N deposition along with deer density and climate was significantly correlated with the main axis of species change while in *Racomitrium* heath there was a less strong relationship between reduced N and species change but a strong relationship with sheep grazing. This suggests that different communities could potentially be sensitive to different forms of N deposition and that the significance of N deposition relative to other drivers varies between communities. Overall however, it appears that N deposition is likely to be a significant factor driving vegetation change in a number of montane communities.

5.1.2 Soil changes

As the sites sampled in this project were not exact replicates of previous samples, it would be inappropriate to draw firm conclusions from this brief analysis. However, we have demonstrated the differences in soil C:N ratios between two of our main montane habitats and that there may be an indication of a decline in soil C:N in the Cairngorms. The results from the Southern Upland *Racomitrium* heaths suggest that despite exposure to relatively large N deposition inputs over 30-40 years there has not been a substantial decline in soil C:N in this habitat.

5.2 Implications

These findings support the idea that N deposition is a significant driver of change in montane habitats but also highlight that fact that climate and grazing have similarly significant effects and must be accounted for when planning management or policy intervention. Across all community types, oxidised N was more significantly related to composition changes than

reduced N inputs, but there is some evidence that this may vary between communities. The observed reduction in between-sample diversity and in the variety of community types represented suggests a reduction in the quality of montane habitats has occurred over the long term for which N deposition is at least partly responsible. The observed slow changes, mainly in the Cairngorms and Southern Uplands, with deposition of 10-20 kg N ha^{-1} y^{-1}, are broadly supportive of current critical loads.

5.3 Outputs

- Vegetation and soil data from the re-survey have been submitted to the UKREATE database.
- 'The only constant is change: using archive data to explore drivers of change in heathland communities over the last 30 years'. Invited talk at 9th National Heathland Conference, University of Aberdeen, August 2006.

6. Possible future work

The 'data archeology' approach used in this study, exploiting the potential of archive datasets as a means of setting up long term comparisons, has shown itself to be both practicable and useful as a means of detecting long term driver impacts. Archive data holdings at Macaulay cover the full range of Scottish vegetation types and there is the potential to do similar work to that described here on a wider range of communities. This study has shown the community-specific nature of changes induced by environmental drivers such as N deposition and highlights the need for community-specific information on driver impacts in order to inform policy development. Moorlands and wetlands are both extremely widespread and extensive habitats in Scotland and are well represented in archive data holdings. Future work on these vegetation types would further enhance our knowledge of vegetation and soil responses to N deposition over the long term allow better validation of modelling studies and enhance predictions of future changes.

7. References

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Work Package 2:
Impacts, Recovery and Processes

Task 10:
Changes in the flora of calcareous grasslands

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Task 10 - Changes in the flora of calcareous grasslands

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1. Summary

This study aimed to address specific gaps in knowledge by assessing whether there is evidence of changes in the species composition of calcareous grasslands over the last 15 years at different sites across the UK that are spatially associated with modeled nitrogen deposition. The work involved a botanical re-survey of a sample of the network of calcareous grassland sites that were established in the early 1990s by Prof. John Rodwell and colleagues of Lancaster University, with support from Defra, to monitor long-term impacts of air pollution and climate change. Bryophyte species composition was surveyed at 16 of the sites in the summer of 2005, and higher plant species composition was surveyed at 25 of the sites in the summer of 2006. There was evidence of an overall decline in vascular plant species richness, and an increase in grass/forb ratio, but this was not related to nitrogen deposition. Vascular plant species richness declined at deposition rates above 20 kg N ha^{-1} yr^{-1} but was also lower at deposition rates below 12 kg N ha^{-1} yr^{-1}. For both bryophyte and vascular plant species, there was a weak, non-significant, positive correlation between change in Ellenberg index and nitrogen deposition. Overall, the results are broadly consistent with the empirical critical load of 20 kg N ha^{-1} yr^{-1} which is currently applied for mapping purposes for calcareous grasslands in the UK, and provide some evidence that changes in species composition and possibly vascular plant species richness may occur in the field at deposition rates above this value.

2. Policy Relevance

Substantial areas of the UK are indicated to be in exceedance of empirical critical loads of nutrient nitrogen, which are largely derived from field manipulation experiments. Evidence that there are real changes in species composition across the UK which are associated with changes in nitrogen deposition is very important in supporting the use of these critical loads, and as evidence that measures to decrease emissions will have significant benefits for habitats of national conservation significance. The research under this Work Package aimed to provide new evidence to support critical loads set for a particular habitat of conservation value, calcareous grasslands.

3. Objectives

This work contributed to two broad objectives:-

- Evaluation of the potential chemical and biological impacts of nitrogen deposition
- Evaluation of the recovery of damaged ecosystems in response to nitrogen deposition reductions and interactions with other variables

This was achieved by a detailed study of species composition change in permanent quadrats receiving different rates of nitrogen deposition. The focus of the second part of the study was

changed from limestone pavement habitats to calcareous grasslands, because of the existence of much better historical information for the latter.

4. Methods and results

The work focused on a national network of permanently marked monitoring plots at 56 calcareous grassland sites, established in 1991-2 and designed to represent the full range of calcareous grassland communities in the UK (Rich *et al*., 1993). In summary, between one and six 12x12 m plots were set up within each site. Each plot was marked out with tapes at 2m intervals and the vegetation was recorded using 36 0.5x0.5 m quadrats. Buried metal coils were used to mark the location of the plots and to facilitate re-location using metal detectors.

The work in 2005 focused, as planned, on the bryophyte flora. The aim was to test whether there is evidence from field sites, with varying rates of N deposition, across the UK, of changes in bryophyte species composition that are consistent with those that we reported at the Wardlow Hay Cop long-term field manipulation experiment under the previous Umbrella contract – specifically changes towards a more nitrophytic and acidophytic bryophyte flora. A total of 16 sites were re-surveyed, giving a range of modeled N deposition from 12.5 to 35.8 kg N ha^{-1} yr^{-1}. Sites were also chosen in order to represent a wide geographical spread and with a dominant National Vegetation Classification of CG2, although CG1 grasslands were included where N deposition was particularly low. Within each of the sixteen sites, one plot was chosen for the re-survey. Other than meeting the NVC requirement, there were no special criteria for choosing the plot. The work in 2006 focussed on species composition of higher plants. The same 16 sites were re-surveyed to obtain data on higher plant species composition, and a further 9 plots were added at the low end of the available gradient of N deposition in the original sites, giving a range down to 5.8 kg N ha^{-1} yr^{-1}. However, this did mean that some sites in CG9, CG10, CG11 and CG13 were included. Soil samples were taken adjacent to the plots for organic content and C:N measurements, and to determine phosphorus, ammonium, nitrate and base cation levels. The results reported here focus on the plant data.

Bryophyte surveys took place in May and June 2005. Relocating the exact location of the original 36 0.5 x 0.5 m quadrats was successful in all plots. A 0.5x0.5 m quadrat was placed as close as was reasonably practicable to the original 36 quadrat locations in turn. All bryophyte species within each quadrat were recorded as presence or absence. Mean soil pH measurements for the whole plot were taken in the field using a portable pH meter with a standard glass electrode at randomly selected points in each of the 36 plots. Vascular plant surveys took place in June, July and August 2006. For the additional nine sites, plots were chosen where prevailing grazing pressure was low. Soil samples were stored at 4 ^{o}C in the dark until further analysis. Soil pH was measured with a pH meter and standard glass electrode at the lab using deionised water for extraction. All vascular species were recorded as presence or absence. Total vegetation cover, bare ground and rock was recorded for every quadrat.

Of the 16 sites surveyed, bryophyte species richness had increased at 11 sites and decreased at 5 sites. A simple linear regression demonstrated no significant associations between species richness and modelled nitrogen deposition or soil surface pH. Of the species that had sufficient points to be able to conduct a simple regression analysis, none exhibited a statistically significant pattern, in terms of frequency change over time, to either mean soil surface pH or modelled nitrogen deposition.

Ellenberg indicator values were obtained from the PLANTATT database and the LEDA Traitbase (Hill et al 2004; Kleyer et al 2006). Ellenberg indicator values for British bryophytes, relating to fertility (*iN*), were applied to each species found at each site and a mean Ellenberg value was calculated for both survey years at each site. From that a weighted average Ellenberg indicator value (iN) was obtained by using the following equation:

$$\mathrm{iN}j = \frac{\sum_{i-1}^{S} (NiXi, j)}{\sum_{i=1}^{S} Xi, j}$$

where
iN*j* = weighted average Ellenberg indicator for site j
Ni = Ellenberg value for species i
Xi,j = number of occurrences of species i in site j
S = number of species

Of the 16 sites surveyed, the average Ellenberg Index values for fertility for bryophytes had increased at 9 (56%) sites and decreased at 7 (44%) sites. However, not all species recorded have an assigned Ellenberg value. A simple linear regression demonstrated no significant (F=2.673; P>0.05) association between changes to the mean Ellenberg Index fertility values and modelled nitrogen deposition. The trend-line shown in Figure 10.1 suggests, however, that a weak nitrogen response may be evident.

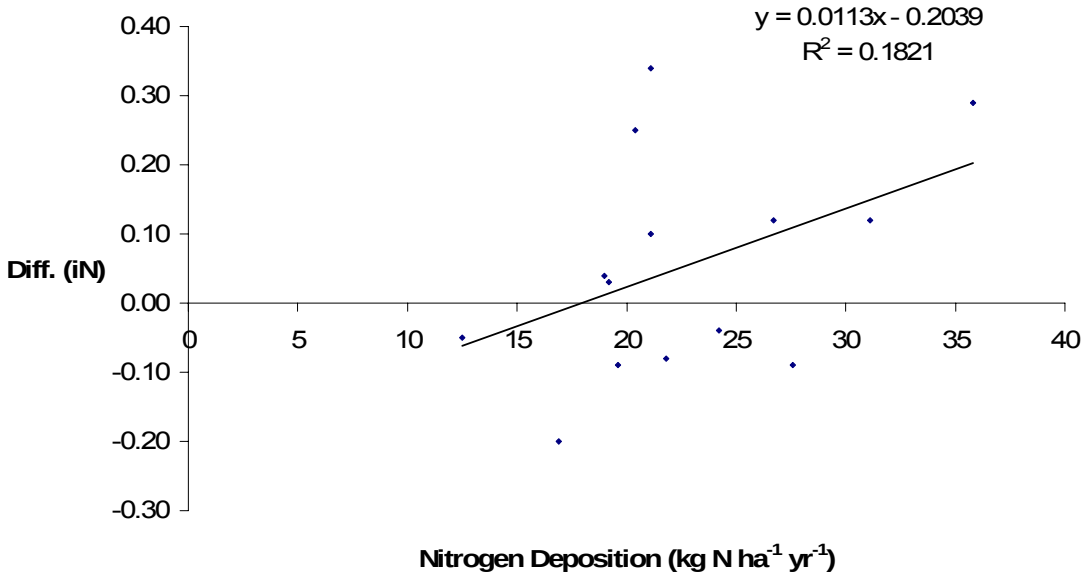

Figure 1. Mean bryophyte Ellenberg (iN) response to N deposition (kg N ha^{-1} yr^{-1}).

In contrast to the bryophyte data, the vascular plant data showed a consistent reduction in species richness between 1991 and 2006. Decreases in species richness were found at 20 of the 25 sites, with small increases found at only 5 sites. The increase in species richness at these 5 sites was a result of an increase in very common, nitrophilic species such as *Arenaria serpyllifolia*, *Brachypodium pinnatum*, *Circium vulgare* and *Trifolium repens*. The change in species richness ranged from an increase of 3 species to a decrease of 18 species. At the majority of sites, there was also evidence of an increase in the ratio of grass to forb species richness, suggesting a selective loss of forb species overall. However, as for bryophytes, there was no evidence that change in vascular plant species richness over the period 1991-2006 was related to N deposition; species richness declined at sites across the full range of N deposition.

However, a very strong curvilinear relationship was found between overall vascular plant species richness and modeled N deposition (R^2 = 0.38; P<0.001), as shown in Figure 10.2. There seem to be two components to the relationship. For sites with current N deposition above 15 kg N ha^{-1} yr^{-1}, there was a strong decline in species richness with increasing N deposition, suggesting that long-term deposition above the UK mapping value of a critical load of 20 kg N ha^{-1} yr^{-1} was associated with the loss of species from these communities. However, sites with a deposition rate below 15 kg N ha^{-1} yr^{-1} also showed lower species richness than those receiving the 'optimal' deposition.

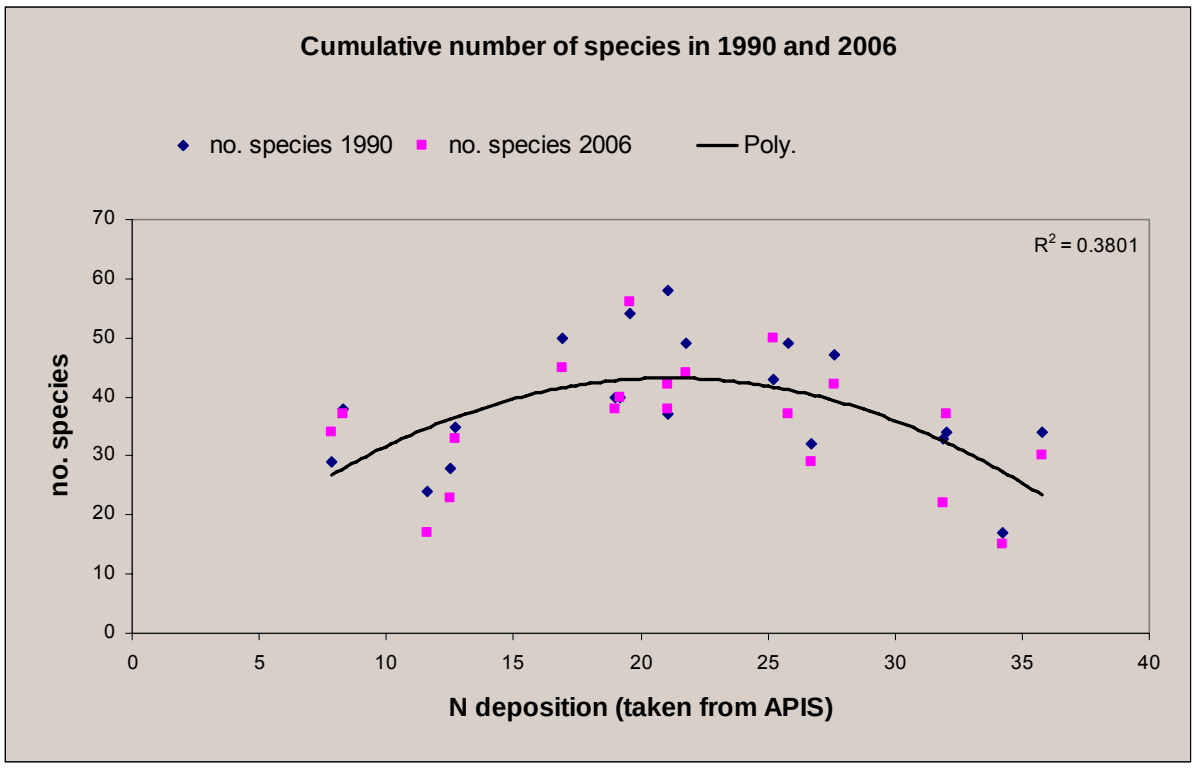

Figure 2. Relationship between vascular species richness and modeled N deposition (kg N ha^{-1} yr^{-1}), including data from 1991 and 2006 surveys.

As for bryophytes, the change in mean Ellenberg index based on the vascular plants for each site showed a trend to increase with increasing nitrogen deposition (Figure 10.3). However, as for bryophytes, the relationship was weak and not statistically significant (R^2=0.25; P=0.057).

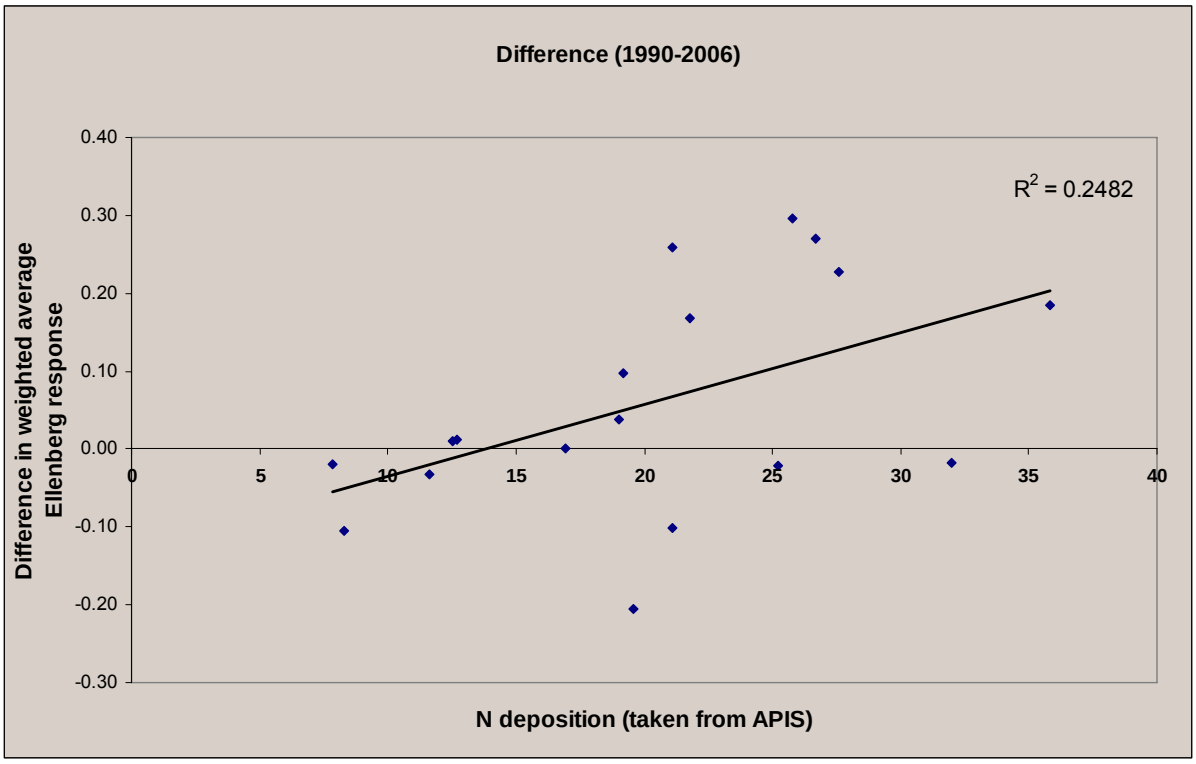

Figure 3. Mean vascular plant Ellenberg (iN) response to N deposition (kg ha^{-1} yr^{-1}).

5. Discussion and implications

The results of this survey need to be considered in the context of the significant limitations of such an exercise, especially with the relatively small number of sites that were examined. On the other hand, the fact that we were able to successfully re-locate all the original plots means that, unlike many similar exercises, we are able to separate spatial variation in community characteristics, such as species richness, from changes in these variables in response to changing N deposition over the last 15 years. The considerable scatter in the data may be explained by a number of factors. Differences in recorder observations between the two surveys may be an issue; for bryophytes, in particular, we have the impression that species may have been under-recorded at some sites in the original survey, thus leading to artificial increases in species richness. The timing of surveys may also be important, as species frequency may be quite variable from year to year. This may be particularly important to the vascular plant survey which was done on average 4 weeks later than the original 1990 survey. Finally, management is not controlled at these sites, and differences in management practice may have a major impact on species richness and composition.

Given these limitations, it is of interest that the results provide some evidence of significant effects of N deposition at sites with deposition rates above 20 kg ha^{-1} yr^{-1}. In particular, there is evidence that vascular plant species richness falls above this deposition rate. Furthermore, some, but not all, of these higher deposition sites show evidence of an increase in Ellenberg fertility index over the last 15 years, for both vascular plants and bryophytes, despite the fact that there has been in general a decline in N deposition rate over the UK in this period. However, the observation is consistent with the fact that, despite this decline in deposition rate, the critical load is still exceeded at these sites.

6. Possible future work

Some further analysis of the data collected within this Work Package will be undertaken later in 2006, using resources within University of York. This will incorporate information about the management of the sites, and how this may have changed over the last 15 years, as this is likely to be an important driver of changes in species composition. In addition, multivariate analysis of the complete dataset will be undertaken to identify particular groups of species which are sensitive to N deposition and to other factors such as management. Once this analysis is complete, it is intended to write up the work for peer-reviewed publication.

There are a total of 56 sites available in the database. The resource limitations within this contract, and the fact that site survey is only possible in specific time windows, meant that less than 50% of these sites could be re-surveyed. It would be extremely valuable to extend the work to cover all the remaining 31 sites. While this would not extend the range of values of N deposition covered, since the 25 sites were selected to cover the full range of values, it would provide a more robust basis for inferences about the extent to which observed differences between sites are causally elated to N deposition. This work will not be taken forward under the next phase of the 'Terrestrial Umbrella' project, and we will therefore seek support for the work from other sources.

Finally, although the network of 56 permanent quadrats was established with Department of the Environment support in the early 1990s, with the aim of providing a basis for long-term monitoring of the impacts of air pollution and climate change in the UK, this is, to our knowledge, the first occasion on which the sites have been re-surveyed. Given the interest within Defra and other agencies in establishing a monitoring network for terrestrial impacts of air pollution and climate change, we highlight the potential value of this network of sites for future impact assessments.

7. References

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Work Package 2:
Impacts, Recovery and Processes

Task 11:
Factors controlling nitrate breakthrough at the landscape scale

M.S. Cresser and S.M. Green

University of York

Task 11 - Factors controlling nitrate breakthrough at the landscape scale

PIs: M. Cresser and S.M. Green

University of York

1. Summary

In our 2006 report, we highlighted the publication of our novel model for predicting spatial and seasonal variation in nitrate concentrations in UK upland rivers, published in Environmental Pollution (Smart et al., 2005). This has been extended, using UK AWMN data sets, to incorporate N deposition as a controlling factor for leaching rate of nitrate in winter and summer so that the model is applicable across the UK, and a paper on this is in course of preparation. The importance of bare rock and slope in these models flags the need to take to take movement of nitrate and ammonium (especially nitrate) transport in water flowing laterally through soils into account when quantifying N critical loads, and the absurdity of not doing so in upland areas if soil N critical loads are based upon vertical N deposition fluxes on a horizontal square metre basis.

The work led us to realize that using water alkalinity to calculate critical loads of surface waters is also conceptually flawed, because current alkalinity has already been used up in part in neutralizing acid inputs, so the current approach is over-protective. We have published a paper in Environmental Pollution demonstrating this (Cresser et al., 2006), based upon extension of our earlier G-BASH model. The paper also demonstrates that runoff amount is crucial to alkalinity concentration across the UK. This highlights a problem with the assumption often made that weathering rate (rather than leaching of base cations) increases with precipitation amount when calculating critical loads. Weathering rate depends upon the rate of production of carbon dioxide in soil, and this is inadequately considered in current methods for calculating soil critical loads.

These and other serious misconceptions that underpin current soil critical loads methodology were discussed by Cresser in his presentation at the Acid Rain 2005 conference in Prague, and this paper has now been published (Cresser, 2007).

Over the past year, analysis of the results of the research by Sophie Green (a 50%-funded DEFRA PhD student on the Umbrella contract with Cresser at York) on the disruption of the N cycle in soils down slope of salted roads has been completed. Five papers have now been submitted emanating from this study (Green and Cresser, 2007 a, b and c; Green et al., 2007 a and b). The main findings are:

a. Runoff from roads subjected to long-term road salting can dramatically raise soil pH down slope in upland areas.
b. The pH increase dramatically changes N cycling in soils down slope, increasing mineralization of organic matter, ammonification, ammonium leaching down slope and nitrification and nitrate leaching.
c. The increase in nitrification substantially increases nitrate leaching to down-slope rivers, and this is readily detectable in field studies.

d. Loss of soil organic matter over decades of salting is so great that organic matter is no longer substantially solubilised by high salt concentrations found in soil solution below road drains.

In a previous DEFRA Umbrella Contract Report and paper (Cresser et al., 2004), we highlighted the fact that ammonium leaching from acid, organic-rich soils to the adjacent river network in the River Etherow catchment in the S. Pennines reflected the slow rate of transfer of ammonium inputs from the atmosphere to soil biomass-N, soil organic matter-N or nitrate. Under these conditions, near-equilibration eventually occurs between soil exchangeable ammonium, effective input concentrations of ammonium from the atmosphere, and ammonium concentration in soil solution (and hence in water draining to streams). A puzzle remaining was why ammonium leaching was still observed under base-flow conditions in summer. Over recent months we have been testing a new hypothesis, namely that ammonium and nitrate leaching occur not just in through flow (laterally draining water), but also down the soil profile to depth. Thus near-surface groundwater components will contain significant concentrations of both ammonium and nitrate that may be mobilized to streams under base flow conditions. These may be supplemented by mineralization of organic matter at depth in the soil profile, especially if the vertical enhanced N inputs reduce the soil C:N ratio at depth in the profile.

The above ideas have been tested with input from two overseas PhD students working with Cresser, Ishaq Mian and Muhammad Riaz, at no cost to DEFRA. Their data so far has confirmed the hypothesis for heavily N-impacted lowland soils under woodland and permanent grassland in the York area, and they will continue to work on this issue in future, and on related hypotheses, but no longer under the auspices of the Umbrella contract.

2. Policy Relevance

The main relevance to policy is that serious consideration needs to be given to modifying methods currently being used in UK uplands to calculate critical loads for planning pollution abatement strategies. In particular, slope processes need to be properly taken into account when valuating N critical loads.

The really very dramatic influence of road salting on N cycling in soils down slope of roads, and the consequential effects on inorganic N and DOC inputs to rivers require further careful consideration in the contexts of the Water Framework Directive and N transport from soils via rivers to the oceans, especially as roads often run parallel to rivers near valley floors over substantial distances.

Careful evaluation is needed of the extent to which N mineralization and N leaching to near-surface ground waters in heavily N-impacted lowland and upland areas poses a potential threat to groundwater quality in the long term and to surface water eutrophication risk in the short and long terms.

3. Objectives

Objectives outlined in the initial proposal were:

- Evaluation of the chemical impacts of N deposition
- Evaluation of the interaction with management
- Assessment of the processes controlling nitrate leaching from soils to freshwaters and the rate of leaching for use in dynamic models

The research set out to answer the question: *What are the key factors controlling nitrate breakthrough and leaching from soils?* It also intended to contribute to answering the questions: *Can indicators be identified for N status of soil/plant systems and N saturation?* and *What indicators are appropriate to gauge critical load exceedence for nutrient nitrogen?* The research objective was be to improve our quantitative understanding of how N species input fluxes interact with other catchment parameters at the landscape scale to influence the capacity of soils to store N inputs, and thus N species leaching (especially nitrate leaching).

4. Methods and results

4.1 Potential disruption of the nitrogen cycle through the application of de-icing salts on upland highways.

4.1.1 Field Site and Methods

The site used is an upland area near the A6 at Selside, Cumbria, UK (NY 554 046 GB Grid), with altitudes up to 458 m above sea level. Soils were podozols with variable depths of organic-rich surface horizons and degrees of podzolisation. Vegetation is primarily grasses, Juncus and bryophytes and the land is used for grazing sheep and cattle. This site was selected as it provides opportunity for 3-impact scenarios, as road drainage is piped directly onto the soil surface at regular intervals. Hence the site presents three possible impact scenarios (a) impacted by direct (channelled from a pipe) drainage plus spray, (b) impacted by indirect drainage and spray and (c) an un-impacted control on an adjacent hillside at the same attitude and aspect within the catchment. A wall ca. 1.5 m high separates the road from the salt-affected sampling sites, but this is down slope and does not represent a significant spray shield.

Sample Collection:
Duplicate transects for each scenario were marked out at a spacing of at least 10 m to avoid pseudoreplication, perpendicular to the road; in the case of the direct (drain affected) transects they were immediately down slope of drainage pipe outlets (T1 & T2); indirect transects (T3 & T4) were to the left of T1 and T2. T5 and T6 refer to the un-impacted control. Soil samples were collected from 0-10 cm with a stainless steel trowel at 2, 4, 8, 16, 32 and 64 m from the wall (i.e initially 4 m from the road), for transects T1 to T4, and from corresponding distances on control transects, on the 7th April 2005. Further soil sampling was completed at 2, 8 and 32 m on the 12th October 2005, when residual salting impacts should be minimal, for inter-seasonal comparison.

Soil Solution Sampling:
Soil solutions for each transect were sampled in triplicate, using MOM Eijkelkamp rhizon soil solution samplers with a length of 10 cm and 2.5 mm external diameter, and 60 ml syringes at 2, 4, 8, 16, 32 and 64 m from the wall on 8 occasions between 12/10/2005 and 08/07/2006. Thus sampling started before any road salt application in the autumn of 2005. Sampling frequency was increased over the salt application period

River Sampling:
Runoff from the 1500-m section of the A6 road investigated drains eventually into Crookdale Brook. Water was sampled along the Crookdale Brook from 12/10/05 to 08/07/2006. Seven locations were sampled (S1-S7) along the Brook where it runs parallel to the A6, at intervals of ca. 200 m. The water was sampled in duplicate.

Soil Analysis:
Soils were analysed for pH, loss on ignition, potassium chloride-extractable ammonium and nitrate, exchangeable base cations and cation exchange capacity (CEC).

Freshwater and Soil Solution Analysis:
Water samples were analysed for base cations, ammonium-N, nitrate-N, organic-N, chloride, fluoride and sulphate using ICP-OES (base cations), ion chromatography (Dionex, anions) and a Bran and Luebe AutoAnalyser 3 (N-species). Performance of all instrumental methods was checked using synthetic reference standard materials. There was no significant difference between nitrate values obtained by ion chromatography and automated colorimetry.

4.1.2 Results from Road Salting Impact Research:

Soil
Figure 1 (upper) shows extractable ammonium in soil in April, at the end of the salting period, as a percentage occupation of CEC at 2, 4, 8, 16, 32 and 64 m for each transect type. The control transects contain far more than the transects for the other pollution scenarios (significant at 1%). In the most salt-impacted soils (T1 & T2) the % ammonium on CEC declines from 2 m to 16 m and then increases with distance from the road (5 % level). This initial decline also was seen for the spray-contaminated transects (T3 & T4) but to 4 m, and the increase in the % ammonium on CEC on moving down slope occurs at a faster rate. The control demonstrates a similar rate of increase as the transect is descended.

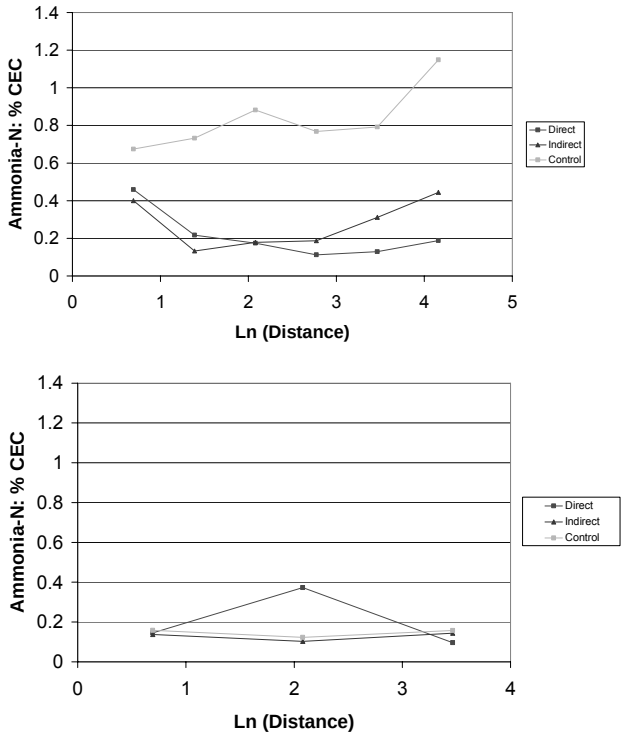

Figure 1. Percentage ammonium occupation of the CEC for April (upper) and October 2005 (lower) at 2, 4, 8, 16, 32 and 64m from the road.

Figure 1 (lower) shows seasonal differences of ammonium % on CEC in October 2005. The less contaminated (spray-affected) and control transects converge to similar values, and the most heavily salt-affected soils at 2 and 32 m also converge to control levels. However, percentage occupation of exchange sites by ammonium remains high at 8 m, so recovery from enhanced ammonium leaching over summer appears incomplete. The percentage occupancy of the CEC by ammonium for all pollution scenarios is lower generally in October than in April.

Figure 2 (left) shows the concentration of extractable nitrate-N for direct, indirect and control soils at 2, 4, 8, 16, 32 and 64 m in April, 2005. The nitrate-N contents of the control soils are significantly lower than those of all salt-affected transects. Spray-affected soils contain the most nitrate (1 % level). Seasonal differences occur between April and October 2005, with more nitrate-N for all scenarios in Autumn, especially for the control.

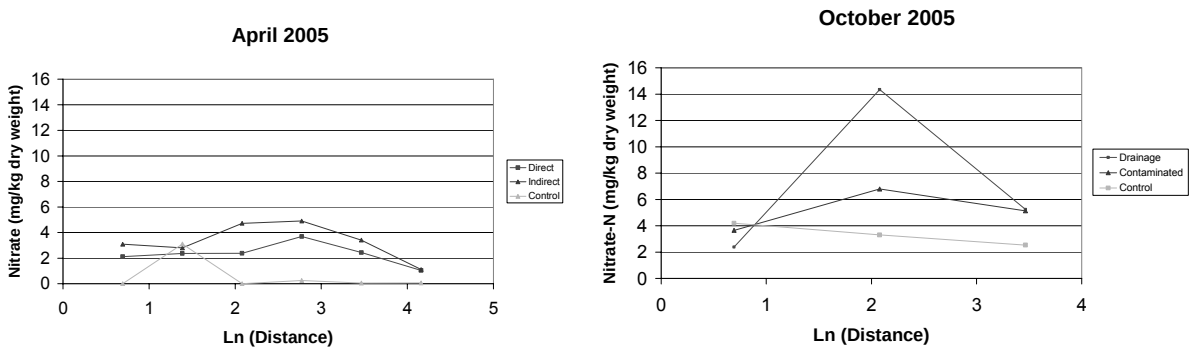

Figure 2. KCl-extractable nitrate-N concentration (mg/kg) for (a) April and (b) October 2005 for soils sampled at 2, 4, 8, 16, 32 and 64 m from the road.

Figure 3 shows how LOI (%) changes with distance from the road in October 2005. Organic matter loss has been very substantial compared to the control soils.

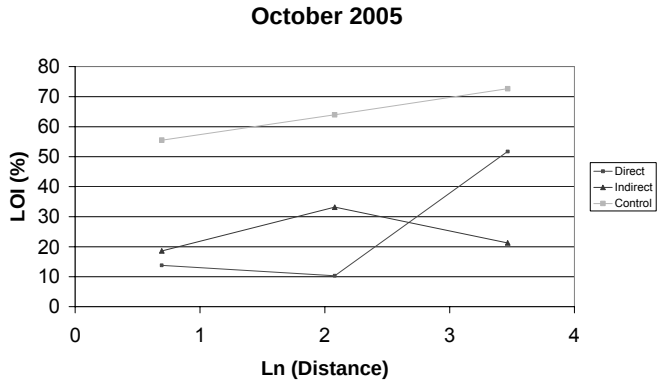

Figure 3. Loss-on-ignition (%) for (October 2005 for soils sampled at 2, 4, 8, 16, 32 and 64 m from the road

Soil pH (H_2O) at 2 - 4 m during April for the drain-impacted transects was more than two units higher than that for the indirectly-effected transects, and two and a half to three units higher than that for the control transects (Fig. 4, left, 1 % level). This corresponds to the high sodium dominance and low hydrogen occupancy of the CEC. The heavily-salt affected soils at 2 m maintain a pH above 7 in October (Fig. 4, right), whilst for the spray-contaminated soils pH has increased by over half a unit compared with the April value.

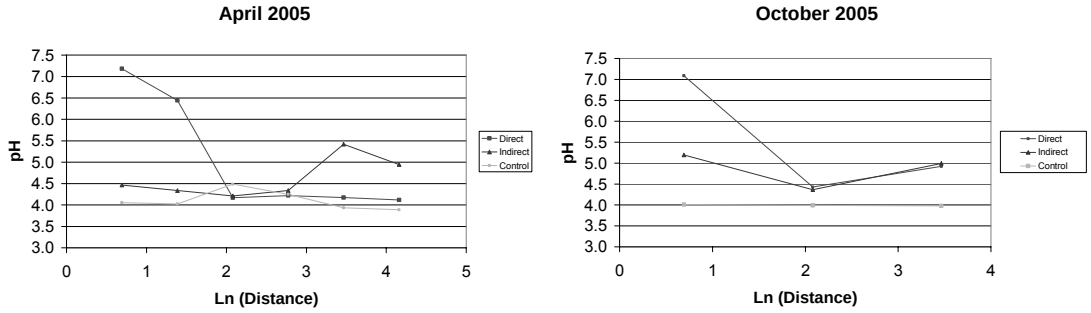

Figure 4. Soil pH (H_2O) for (a) April and (b) October 2005 for soil samples at 2, 4, 8, 16, 32 and 64 m from the road.

Soil Solution:

Soil solutions sampled from 12th October 2005 to the 8th July 2006 highlighted high concentrations of sodium declining with distance from the road. Concentrations up to 6000 mg/l were recorded 2 m from the road at drain-affected sites. In comparison, for the spray-affected transects soil solutions contained only ~150 mg/l sodium at 2 m, declining to 15 - 18 and 15 -.25 at 64 m; the controls had background sodium concentrations of 2 - 5 mg/l across all distances even on 03/01/06. The trends in sodium concentration were reflected in chloride data. The large episodic increases in sodium and chloride concentrations were closely associated with road salt applications dates throughout the winter, while data from 12/10/05

and 03/11/06 provide background soil solution concentrations prior to initiation of salt application for winter maintenance.
The control soil solutions very consistently contained minimal amounts of nitrate-N, whilst the salt-affected transects had spatially and temporally variable concentrations. Ammonium concentrations tended to be higher in soil solution closer to the road during the period that road salts were being applied to the highway in December and January for T1 and T2 (drain affected) as well as for T4 (spray-affected), although T3 did not show such a trend.

Crookdale Brook:
The stream water analyses showed that water pH and concentrations of the base cations Ca, Mg and Na in the river increased as the road and Crookdale Brook converged. However base cation concentrations varied over time too, and were especially high during periods when winter maintenance was prominent (22/11/05- 27/01/2006), suggesting relationships between the quantities of base cations in Crookdale Brook and road salt application.

The sampling sites ranged over 1.5 km of the water course from S1, where there was no road salt effect, to S7, with a progressive increase in the road effect. Site S7 might be expected to have the greatest potential affect as it is closest to the road, with a drainage pipe entering the system up slope, just prior to the river sampling point. However, the concentration of sodium ions at S7 is diluted by the introduction of additional water from the Borrowdale Brook tributary prior to the sampling point. Up stream of S1, the catchment area spans over ca. 7 km^2; hence, the enhancements in base cation concentrations found were very significant, bearing in mind the relative area of unaffected catchment upstream of S2. A considerable road impact occurs within a few hundred metres. This trend was also apparent in chloride concentration, and there were significant positive relationships between the concentrations of base cations (Na, Mg and Ca) and chloride (all at the 1 % level).

Figure 5 shows that nitrate-N in the river increased steadily as the road and Crookdale Brook converged, especially during periods when winter maintenance was prominent (22/11/05-27/01/2006). There is a relationship between the concentrations of nitrate and sodium observed (5 % level) There did not appear to be a spatial trend with respect to the river ammonium concentrations.

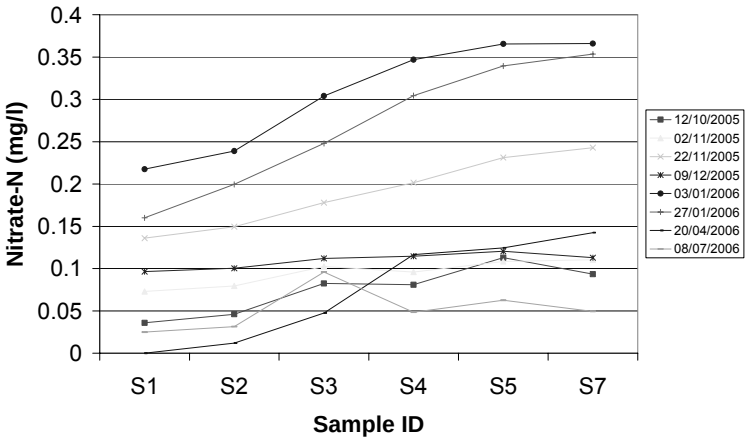

Figure 5. The change in nitrate-N concentration (mg/l) with time and distance along Crookdale Brook for the period 12/10/05 – 08/07/2006.

4.1.3 Discussion

Organic matter is mobilized due to a loss of soil aggregate stability following displacement of calcium and magnesium ions from soil organic matter cation exchange sites by sodium ions, although the road salt used in Cumbria contains a small amount of gypsum. From Fig. 3 it is clear that organic matter loss from salt-impacted soils has occurred. This loss would be facilitated by the higher soil pH in the salt-effected soils (Fig. 4), especially at 2 – 4 m. There would be less organic matter input near drain-affected areas too which, coupled to past heavy leaching loss, could lead to maximum accumulation at 32 m.

We have carried out additional experiments in which we extracted roadside soils subjected to diverse salt impact scenarios in the field (over decades) with environmentally relevant salt concentrations, and examined variations in DOC concentrations down transects. These very clearly confirmed that most of the organic matter that was potentially mobile has already been dispersed close to the road (Green and Cresser, 2007c; Green *et al*., 2007b), so evidence of such facilitated movement in soil solution now is unlikely as road salting has been conducted in this region for several decades. However in the past, enhanced dispersal of organic matter, which is still seen for control soils today, would have also contributed to the loss of soil organic matter and hence the lower LOI% in the salt-impacted soils.

The interpretation of the possible influence of road salting on the nitrogen cycle is complex due to the number of possible drivers involved and effected by salting, such as pH (Fig. 4), salinity and the displacement of organic matter (including organic-N) on key processes in the nitrogen cycle such as ammonification and nitrification, as well as complex cation exchange effects. It has been shown clearly though that ammonium % occupancy of the CEC has effectively been reduced over time in the salt-affected transects, by displacement and/or enhanced nitrification at higher soil pH. Hence, a major displacement flush is unlikely to be observed in the soil solution data in this study. However, there is a initial increase in ammonium-N in soil solution at 2 m; thus some displacement may be occurring or a road runoff influence is present.

The reduction in CEC has a knock-on effect on bioavailability of ammonia for further processes or direct plant uptake. What also needs to be considered is the role of salinity and pH shifts on the microbial population as well as effects upon soil structure stability and the enhanced organic matter mobility; these together ultimately affect function. Some residual salinity effect and other possible effects cannot be ruled out as possible influences on the microbial population; however, they must be small compared to the acidity neutralisation effect on microbial activity. Long-term soil pH effects are clearly very important to nitrification. The greater acidity in control soils inhibited nitrification, or rather the increase in soil pH caused by the salting impact greatly favours nitrification.

There are clear spatial variations between and within transects with respect to pH that will induce nitrification enhancement or suppression within the study area. Elevated nitrate-N concentrations above control levels were evident in both drainage and spray-affected soils, suggesting that nitrification has been enhanced by the elevated pH. Soil solution nitrate-N concentrations are elevated above the levels observed in the control soils, which also indicates enhanced potential for leaching to the Crookdale Brook (Fig. 5). Extractable nitrate-N concentrations in soils in April were generally less than those in October (Fig. 2) due to greater microbial and/or plant uptake of the nutrient due to growth induced in Spring.

There is clearly potential for enhanced nitrate loading of waterways due to leaching, which has particular relevance to the Water Framework Directive. This is especially true in the UK uplands where major trunk roads often run parallel to rivers that supply a substantial portion of UK potable waters. Yet it is in these areas where de-icing salts are most likely to be needed to maintain the flow of traffic through the winter months on roads and have been in continuous use for tens of years, and de-icing salts have been used as a winter maintenance practice for as long.

4.2 Do Nitrate and Ammonium in Rivers Originate at Depth in Soil Profiles at Heavily N-impacted Sites?

4.2.1 Field Site and Methods

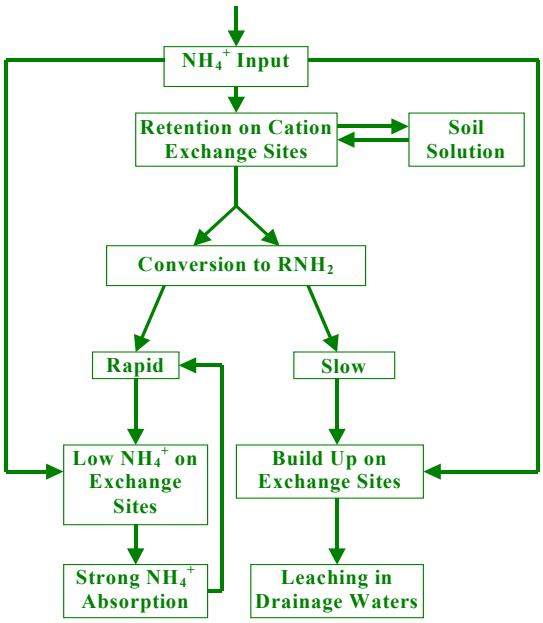

Figure 6. Flow chart to show conditions under which ammonium leaching in/from soils occurs in heavily N-impacted areas.

Based upon Fig. 6 from Cresser et al. (2004), which explains how a low rate of conversion of ammonium to nitrate or organic N in heavily N-impacted area may lead to ammonium leaching from soils, it seems likely that in N-polluted areas, ammonium will move down soil profiles. If ammonium can move through soil to rivers, it can move down the soil profile too. We need to know:

- If ammonium leaches down soil profiles.
- If it is nitrified at depth.
- If it is produced at depth.
- If it is retained at depth, and if so how/where?
- What controls these processes?

We have been using soils from 7 profiles at Hob Moor, a nature reserve managed as a low nutrient status permanent grassland but with a small area of deciduous woodland, just south of the city of York to start to try to answer these questions.

- Samples were collected at 0-15, 15-30, 30-45 and 45-60 cm depths.
- They were analysed for pH, moisture content, P status, %C, %N, C:N ratio and KCl-extractable ammonium-N and nitrate-N.

- In addition field moist samples from each depth and each profile were spiked with known amounts of ammonium to see how much nitrification occurred over 7 days.
- The spiked samples were extracted with 1M KCl and extractable ammonium-N and nitrate-N were again measured.

4.2.2 Results

For sites 1 – 4, soil pH increased with depth, as would be expected for minimally managed grassland (no liming) in a fairly heavily S & N polluted environment like York over recent decades. Soil profiles acidify from the top downwards. Duplicates show good agreement.

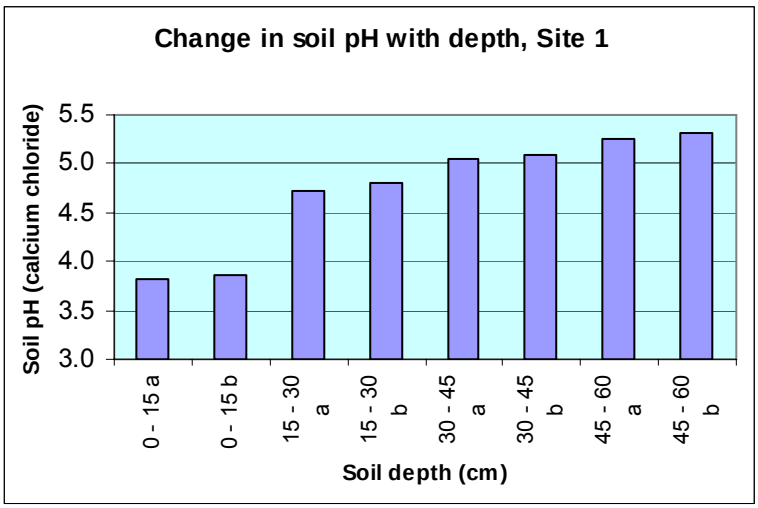

Figure 7. Typical results for soil pH vs. depth for freely draining soils at Hob Moor.

Surprisingly, however, for sites 5 – 7, soil pH decreases or changes irregularly with depth, suggesting possible liming in the distant past, or deposition of parent material of higher base cation status towards the surface over material of lower base status.

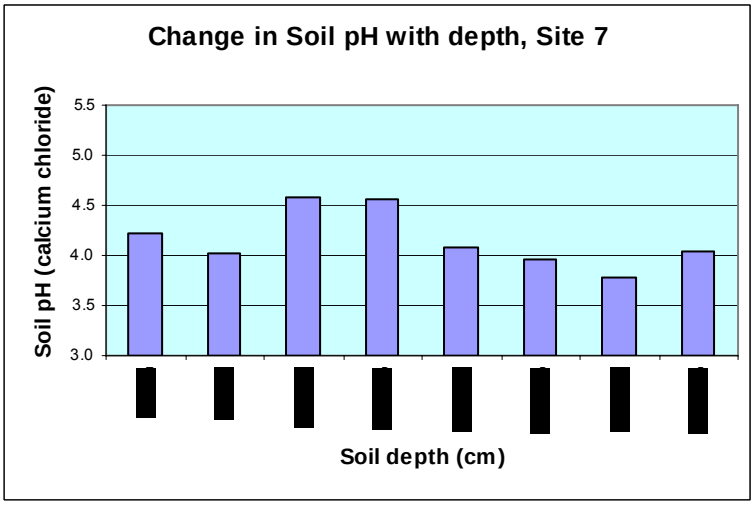

Figure 8. Change in pH with soil depth for Site 7 at Hob Moor

The extractable ammonium-N declined with depth, but nitrate increased. The increased nitrate could be due to nitrate leaching from surface horizons, or ammonium leaching followed by nitrification, or mineralization of organic matter at depth via ammonium to nitrate. Duplicates again show quite good agreement. Fig. 9 shows typical results.

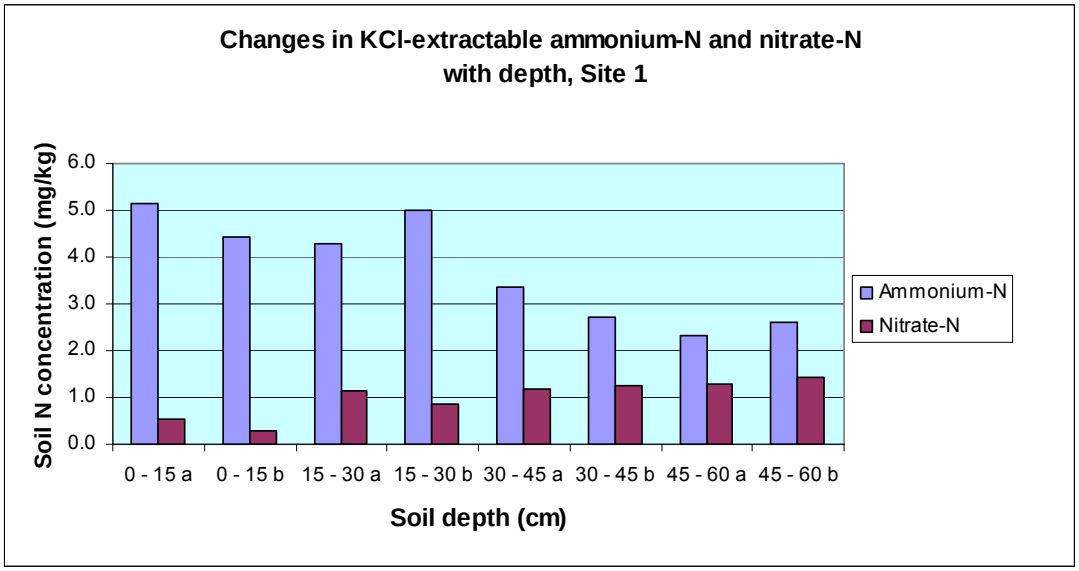

Figure 9. Variation in extractable ammonium-N and nitrate-N with depth at Site 1 at Hob Moor.

For site 1, there is still plenty of organic C and N at depth, so mineralization is probable for in-situ production of ammonium and nitrate. Below 30 cm depth pH is > 5, making mineralization and nitrification more probable. The spiking experiment should show if nitrification will occur. The controls should show how rapidly any ammonification occurs. In this respect Site 1 is quite typical of the results for all 7 sites, although sites 4-7 had higher organic matter contents at depth.

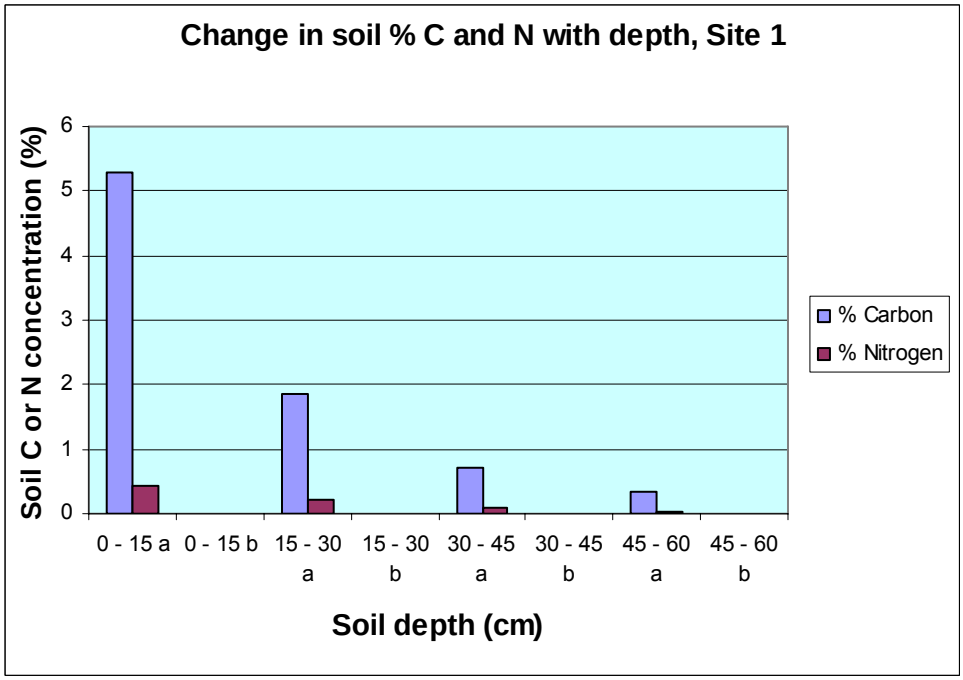

Figure 10. Change in soil C% and N% with depth, Site 1 at Hob Moor

However, Sites 2, 3, 4 and 7 showed an increase in ammonium at 45 - 60 cm depth compared to 30 – 45 cm depth. We need to know if this is from in-situ ammonium production or from ammonium leaching down the soil profile. Ammonification seems most probable, as the CEC will be lower when the organic matter content is much lower at depth. This means that the % of the exchange sites occupied by ammonium must be higher at 45 – 60 cm than at 30 – 45 cm.

For all sites, C:N ratio declines systematically with depth, often to low values, probably reflecting the high in-situ mineral N production and mineral N leaching down the profile in this high N deposition area (Fig. 11).

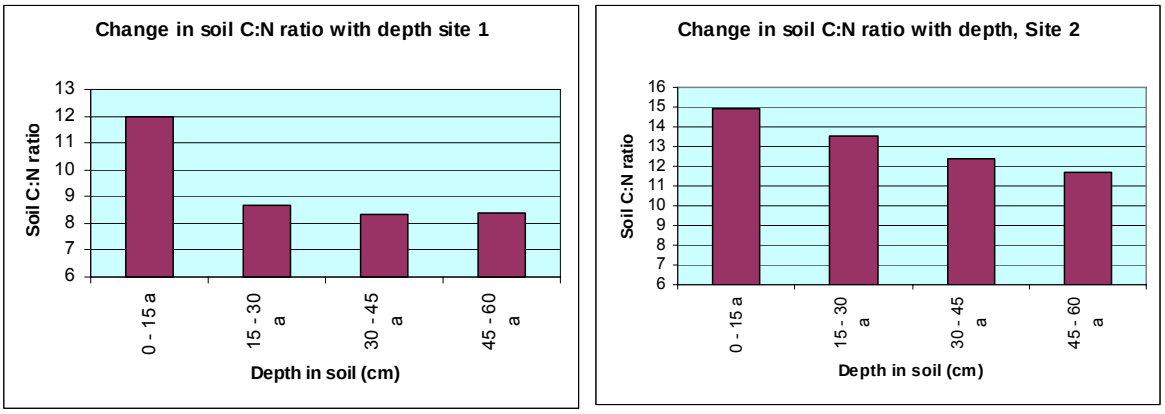

Figure 11. Examples of the change in soil C:N ratio with depth, for Hob Moor Sites 1 and 2.

4.2.3 Discussion of Hob Moor data

The above results show that at Hob Moor, significant concentrations of both ammonium and nitrate occur well below the rooting zone. These would be available to leach into the adjacent becks at the edge of Hob Moor, and down to ground water. Our subsequent ammonium spiking/incubation experiments have confirmed that both ammonium and nitrate are being produced in situ in the soil profile at 45 – 60 cm depth. The in-situ production is however being added to by leaching from higher in the soil profile. Thus the very low C:N ratios at depth are hardly surprising, and are a cuse for concern because clearly they facilitate nitrate and ammonium mobility compared with interim retention in soil microbial biomass.

Acknowledgements:
I am grateful to the second year Environmental Science students from the 2006-7 academic year at York for digging the soil pits at Hob Moor and patient hand sorting to remove roots and stones so that the field moist samples could be sub-sampled for analysis within 2 hours of collection.

5. Discussion and implications

It is clear that the disruption of N cycling by road salting is far from trivial. It needs to be taken into account in several contexts, including enhanced leaching of N species into surface waters, changes in DOC concentrations entering surface waters, and possible impacts upon

local biodiversity. It must be more seriously considered in studies that have attempted to assess impacts of roadside NOx and NHy emissions on roadside biota. This work has attracted considerable interest in the UK, in the USA and in Australia, and as a result the PI now has an on-going research link with the Civil Engineering Department at Monash University. Via this link he has been developing new models for the efficiency of biofilters for N removal, based upon a new understanding of what happens during soil drying and rewetting cycles.

It is also clear from the most recent work that leaching of N species down the soil profile and in-situ production of both ammonium and nitrate in sub-soils will contribute to elevated inorganic N species concentrations in surface waters under base-flow and high-flow (via the groundwater end member) conditions, and may well explain the concentrations found in rivers even during periods of optimal growth during summer. It is also clear that these processes must be adding to groundwater contamination. However additional studies are needed in low N deposition areas before the full significance of the findings can be assessed.

We are closer now to being able to improve our models for predicting surface water chemistry across the UK, taking effects of N and S deposition into account. The PI for Task 11 has a collaborative research link with the Forest Research Institute in Athens, and joint field work with that institute has exposed problems in the recovery of forest soils, in part due to liberation of protons during sulphate desorption during the recovery phase from S deposition.

6. Possible future work

The PI for task 11 will no longer be part of the Terrestrial Umbrella contract. However he expects to have between 3 and 5 PhD students working on soil N cycling over the next 2 to 3 years. They will extend the work on subsurface process that function as sources and sinks of inorganic N species, and on the environmental relevance of these processes at the landscape scale. They will also consider more sensible ways for quantifying N critical loads for soils than those currently employed and methodology for quantifying current soil-N pollution status.

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Work Package 2:

Impacts, Recovery and Processes

Task 12:

Investigation of long-term soil and water dynamics in sensitive Lake District catchments

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Task 12 - Investigation of long-term soil and water dynamics in sensitive Lake District catchments

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1. Summary

During the last 50 years nitrate concentrations in Buttermere and Wastwater (Cumbria, UK) have risen significantly, by 70 and 100% respectively. By estimating contemporary nitrate fluxes in the lakes' catchments and in sub-catchments and comparing them with the fractional areas of different soil types, it is deduced that the surface water nitrate is derived almost entirely from organic-rich ranker soils that have a limited ability to retain atmospherically-deposited nitrogen. Little or no nitrate leaches from the other major soil type, a brown podzol, despite it having a lower C:N ratio (12.0 g g^{-1}) than the ranker (17.0 g g^{-1}), nor is there much contribution from the small areas of improved (chemically fertilised) grassland within the catchments. Although some nitrate leaching is occurring, total N losses are appreciably smaller than atmospheric inputs, so the catchment soils are currently accumulating between 3 and 4 gN m^{-2} a^{-1}.

2. Policy Relevance

The results have established the necessary baseline for continued soil and water monitoring of nitrogen build up in these sensitive catchments, which is necessary to demonstrate the effects of emission control policy. Furthermore, the success of the policy may depend substantially on which of two possible mechanisms of nitrate leaching (incomplete uptake or N-saturation) is operating in the study catchments. The mechanisms can be distinguished by the convincing parameterisation of models that describe both long-term, consistent, trends and short-term variability. The present data should make a significant contribution to that parameterisation

3. Objectives

- Evaluation of the chemical impacts of N deposition
- Evaluation of the interaction with management
- Assessment of the processes controlling nitrate leaching from soils to freshwaters and the rate of leaching for use in dynamic models

4. Methods and results

4.1 Study sites

The study focused on two lakes, Buttermere and Wastwater (Table 1). The catchment of Buttermere is underlain by metamorphosed igneous Ordovician rocks of the Borrowdale Volcanic Group and by sedimentary mudstones, siltstones and sandstones of the Skiddaw Group. The Wastwater catchment is underlain mainly by Borrowdale Volcanic rocks. There

are also some glacial drift deposits. Two main soil types, humic rankers and brown podzolic soils, together account for 80-90% of the catchments' land area. The main land use in the area as a whole is sheep grazing on unimproved pasture. Both lake waters are soft and circumneutral. The average temperature of the catchments is 7°C, the annual rainfall 2710 mm, and the annual evaporation rate 496 mm. The average annual total N deposition during the period 1989 - 2005 was *ca.* 3.5 g m^{-2}, or 35 kg ha^{-1}.

Table 1. **Lake and catchment characteristics.** The data are from Ramsbottom (1976), NATMAP vector soils data (National Soil Resources Institute, 2001), CEH Land Cover Map 2000 (Fuller et al, 2002), EDX Terrain Database for Great Britain.

	Buttermere	Wastwater
Location	54° 31' N 3°16' W	54° 28' N 3°18' W
Lake volume, m^3 x10^6	15.2	116
Lake area, m^2 x10^6	0.94	2.91
Land catchment area, m^2 x10^6	17.8	39.7
Mean catchment altitude, m	380	385
Mean residence time, yr	0.37	1.23
Land cover, %		
acid grassland	42	32
bracken	39	51
shrub heath	8	4
woodland	5	2
improved grassland	4	4
Other	2	7
Soil types, % of land area		
rankers	54	70
podzols	46	30
Soil types mean altitudes, m		
rankers	527	497
Podzols	231	197

4.2 Methods

Lake nitrate concentration data from 1955 to 2005 were assembled from publications and contract reports describing work performed at the Ferry House, Ambleside (Freshwater Biological Association, Institute of Freshwater Ecology, Centre for Ecology and Hydrology). Account was taken of the different methodologies; phenoldisulphonic acid method (1950s), spectrophotometric measurement of nitrite following reduction by cadmium amalgam and diazotization (1970s), ion chromatography (1980s onwards).

Seven streams in the lake catchments were sampled at two-weekly intervals, and analysed for acid anions, including NO_3, by ion chromatography using a Dionex DX-100 instrument. Soils were sampled on non-improved land in the catchments of Scoat Tarn (5 locations), Lingmell Beck (6), Sail Beck (6) and Gatesgarthdale Beck (5), by digging small pits down to bedrock, or to where the profile consisted almost entirely of stones. Horizon bulk densities

were determined, taking account of stones and rocks. The C and N contents of the < 2mm fraction were determined with a Universal CHNS-O Vario EL elemental analyzer, and combined with the bulk density data to estimate soil C and N pools.

The percentage areas of different soil types and their associated altitude ranges were calculated for each catchment within a GIS, using the NATMAP vector soil data (National Soil Resources Institute, 2001). Two main soil types, humic rankers and brown podzolic soils, together account for 80-90% of the land area in the catchments.

4.3 Results

The streams displayed a seasonal trend in nitrate concentration, with low concentrations in summer, attributed to greater use of N in plant growth. Average nitrate concentrations of streams entering the lakes in 2005-6 ranged from 4.4 to 24.3 μM, a similar range to those reported for other upland streams in the Lake District and at other upland UK sites. Within the Wastwater catchment, Lingmell Beck and Mosedale Beck were sampled above the main area of improved grassland, and their combined area-weighted average nitrate concentration was determined to be 22.3 μM. Their confluence, the main inflow to the lake, was sampled after passing through the improved grassland, and its average concentration was 24.3 μM.

Nitrate concentrations in the lakes as a function of time are plotted in Fig. 1. The average rate of increase in nitrate concentration in Buttermere has been 0.14 μM yr^{-1}, while that in Wastwater was 0.27 μM yr^{-1}. The numerous data for Buttermere reveal considerable seasonal and interannual variability.

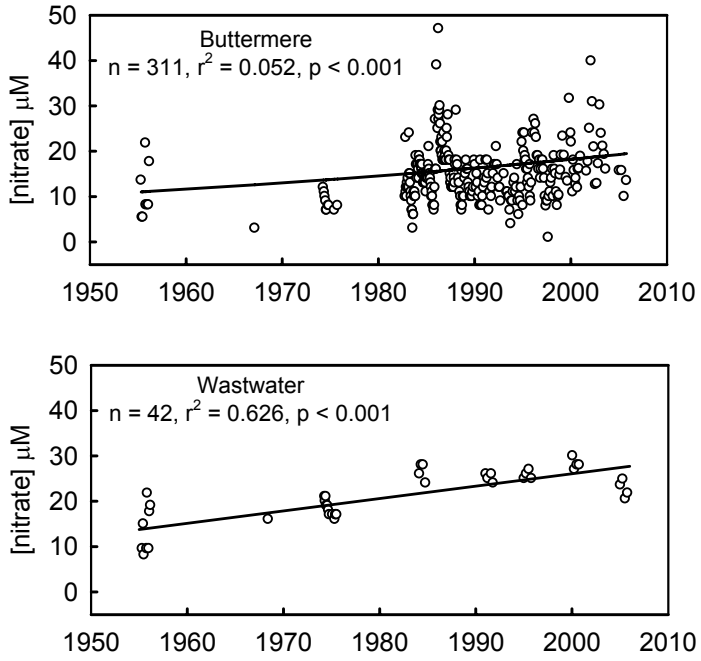

Figure 1. Lake water nitrate concentrations, 1955-2005. The trend line for Buttermere is from the regression of $\log_{10}$ $[NO_3]$ against time. The trend line for Wastwater is the linear regression.

Twenty two soil profiles in areas not containing improved grassland were sampled and the C and N pools (g m^{-2}) quantified (Table 2). The sampled podzols were at lower altitudes, in agreement with soil survey data (Table 1). The podzols have less carbon than the rankers, but the difference in N pools is much less. Thus, the C:N ratio for the whole profile is significantly lower in the podzols. There are only small variations in C:N ratio with horizon. We estimated contemporary nitrate-N fluxes within the 9 stream and lake catchments from our own data, and also using data from the UK Acid Waters Monitoring Network for Scoat Tarn, which lies within the Wastwater catchment. Daily runoff for an individual catchment was estimated from rainfall at Ambleside corrected for altitude, together with monthly-averaged MORECS evaporation rates. For stream waters, daily nitrate concentrations were estimated by linear interpolation of measured values. Daily fluxes (gNO_3-N m^{-2}) were calculated as the product of the runoff and concentration values. Input nitrate fluxes needed to maintain the observed lake water nitrate concentrations were calculated, again using estimated daily runoff, taking lake volume into account, and making corrections for direct nitrate inputs to the lake surface from atmospheric deposition (5-10% of the total), and the evaporation of water from the lake surface. The estimated fluxes are plotted against the percentage area of ranker soil in the stream or lake catchment in Fig. 2. Normal linear regression of the data gave a negative nitrate flux at zero % ranker, which is impossible, and therefore the regression line was forced through the origin. Soil type accounts for 52% ($p < 0.02$) of the variance in the flux.

Table 2. **Soil data.** Values in brackets are standard errors.

	Rankers		Podzols	
no of samples	11		11	
mean altitude, m	586 (43)		301 (40)	
By horizon				
	Oh	Ah	A	B
thickness, cm	9.0 (1.3)	15.0 (2.2)	8.1 (1.1)	28.0 (3.6)
%C	36.1 (2.4)	19.4 (4.9)	12.3 (1.6)	6.5 (1.6)
%N	2.28 (0.15)	1.18 (0.26)	0.91 (0.09)	0.51 (0.09)
C:N g g^{-1}	16.0 (0.7)	15.6 (0.9)	13.4 (0.5)	11.7 (0.9)
pH	4.2 (0.1)	4.4 (0.1)	4.4 (0.08)	4.7 (0.1)
Whole profile				
thickness, cm	22.5 (2.6)		37.3 (3.6)	
%C	26.0 (3.8)		6.9 (0.9)	
%N	1.54 (0.18)		0.56 (0.05)	
C:N g g^{-1}	16.7 (0.9)		12.0 (0.6)	
C kg m^{-2}	11.2 (1.5)		7.8 (1.6)	
N kg m^{-2}	0.70 (0.09)		0.63 (0.10)	

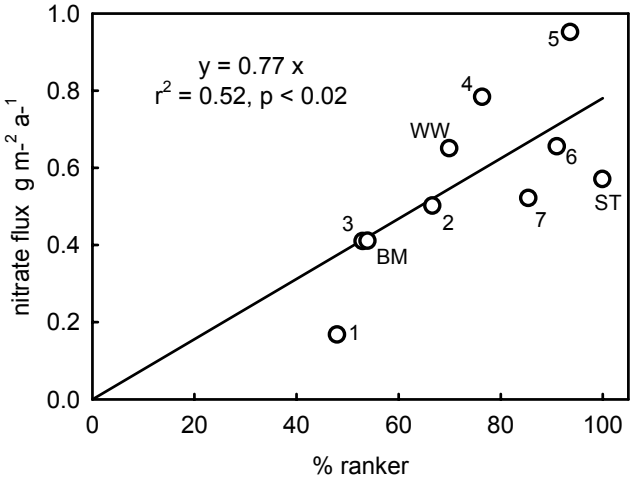

Figure 2. Variation of estimated nitrate leaching fluxes in streams and lakes with the areal percentage of ranker soils in their catchments. The numbers refer to streams, and the letters identify the lakes (BM Buttermere, ST Scoat Tarn, WW Wastwater).

5. Discussion and implications

The data in Figure 1 provide strong evidence of long-term rises in nitrate concentrations in both Buttermere (70% increase) and Wastwater (100% increase) over the last 50 years. For Buttermere, the trend is obscured by the large seasonal and interannual variations, but because Wastwater has a much longer residence time, seasonal variability is not evident in its nitrate concentrations, and some of the interannual variation will also be dampened. Although the increases in nitrate concentration are statistically highly significant over the 50 year period, if only the last 20 years are considered then no significant trends are found.

The period of the nitrate records for the two lakes coincides with the start and continuation of the application of chemical fertilisers to improve grazing land within the Lake District, but it is unlikely that fertiliser nitrogen has contributed significantly to the increased nitrate levels in the two lakes, for the following reasons. (a) Nitrate leaching rates from Lake District catchments dominated by improved grassland are in the range 0.5 – 1 gN m^{-2} a^{-1}, i.e. at the lower end of the range considered above; from these figures, the contributions of fertiliser N to the nitrate increases in Buttermere and Wastwater are only 18% and 11% respectively. (b) Nitrate concentrations in the main Wastwater inflow are very similar to those in the two main contributing tributaries, sampled above the main area of improved (fertilised) grassland within the catchment, which indicates that the streamwater is not acquiring additional nitrate during its passage through the fertilised zone. (c) There is no relationship between the nitrate-N fluxes and the fractional areas of improved grassland. Therefore the most likely explanation for the increases in the lakewater nitrate concentrations is increased leaching from the catchments as a whole, rather than from the small areas of improved grassland.

The results in Figure 2 show that differences in soil type can explain 52% of the variance in the fluxes. The regression indicates that there is essentially no release of nitrate from the podzols, while the average leaching rate from the rankers is 0.77 gN m^{-2} a^{-1}. The areas of ranker soils occur in the higher parts of the catchments (Tables 1 and 2), and so the plant-soil systems receive greater deposition and experience lower temperatures. Moreover, the slopes are greater than in the areas underlain by podzols. All these features have been associated with nitrate leaching, because the lower temperature and shorter growing season make vegetation less demanding of N, and because fast water movement through the soil reduces the possibility of nitrate-N uptake by roots and microbes, especially during winter.

A second possible explanation for the limited ability of the ranker soils to retain N is that they are nitrogen-saturated. Thus, the plants and microbes might be able to take up all incoming nitrogen, but the soil store may have reached the point where it is unable to retain the accumulated element. If so, then the ability of the podzols to retain all incoming N at a C:N ratio of 12.0, while the rankers are apparently N-saturated at a higher C:N ratio of 16.7, needs to be explained; there may be differences in organic matter quality between the soils.

We estimated nitrogen inputs and outputs, and thereby the soil accumulation rate of N. Solution losses of N, apart from nitrate losses, are as ammonium, concentrations of which are generally less than 1 μM in upland Lake District waters, and dissolved organic N, which had an average concentration of *ca.* 5 μM in streams of the Netherbeck catchment in 2001-2002 (R.P. Smart and M. S. Cresser, pers. comm.). Concentrations of particulate N are *ca.* 1 μM. Combining these concentrations with a run-off of 2200 mm gives non-nitrate streamwater losses of approximately 0.2 gN m^{-2} a^{-1}. The rate of removal of N in livestock is less than 0.1 gN m^{-2} a^{-1}. Gaseous losses of N_2O and N_2 are no more than 0.2 gN m^{-2} a^{-1}. Thus, the total rates of loss of N are 0.2–0.4 and 1.0–1.2 gN m^{-2} a^{-1} for the podzols and rankers respectively. The depositional input of N is estimated to be 3.5 gN m^{-2} a^{-1}, to which can be added inputs due to N fixation, estimated at 0.5 to 1 gN m^{-2} a^{-1}. Therefore the catchments are net sinks of nitrogen, which is currently accumulating at rates between 3 and 4 gN m^{-2} a^{-1}, equivalent to an annual increase in the soil N pool of about 0.5%.

This work is described and discussed in more detail in Tipping et al. (2007).

6. Possible future work

The data obtained in the present study provide a long-term perspective on nitrogen behaviour in the uplands of the UK, and can contribute significantly to modelling efforts, in which a key objective is to explain both long-term nitrogen accumulation and short-term seasonal and interannual variability in nitrogen pools and fluxes. It is especially important to distinguish between the two possible mechanisms of nitrate leaching identified in Section 5, since, as was observed by Evans et al. (2004), the "incomplete uptake" mechanism could imply a sustainable long-term sink, whereas N-saturation would imply continued increases in nitrate leaching. The detailed time series data for Buttermere (Figure 1) and other lakes would help to constrain nitrogen cycling models. To generalize the findings, the modelling efforts would benefit from studies of N processing in contrasting soils from different locations, to identify the soil properties that control N uptake and release. Such work could combine the determination of widely-applicable attributes (vegetation type, soil N and C pools) with measurements of C turnover rates (using ^{14}C), mineralizable N, and the fate of added ^{15}N in the rooting zone.

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Work Package 2:

Impacts, Recovery and Processes

Task: 13

Wetlands as nitrate regulators

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Task 13 - Wetlands as nitrate regulators

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1. Summary

The purpose of this research was to examine the 'nitrogen function' of freshwater wetlands typical of near N saturated UK uplands. Upland wetlands occupy areas of marginal agricultural land that are commonly of low economic value, being mostly managed for livestock rough grazing or the maintenance of Red grouse populations for shooting. These environments are also characterized by near N saturation and high rainfall. The landscape significance of upland wetlands that include ombrotrophic peatland, rheotrophic flushes and mineratrophic riparian mires is twofold. First, they are valued reservoirs of biodiversity that contain rare species of plants and invertebrates that support unique bird and wildlife populations. Second, they exist at the confluence of hydrological and biogeochemical processes that together regulate catchment nitrogen dynamics including the retention or release of NH_4^+ and NO_3^-. In this work a ^{15}N pool dilution tracer approach was used to examine gross NH_4^+ and NO_3^- fluxes in three distinct upland wetland types (ombrotrophic peat, mineratrophic flush and rheotrophic riparian mire), seasonally (winter, spring, summer, autumn) in Plynlimon Wales. The results of this work show that all of the wetlands were net NO_3^- exchangers with the rheotrophic flush being the least active. Despite high gross fluxes, net NH_4^+ exchange within the wetland soils was small. There was significant nitrogen retention (microbial immobilization) in the mineratrophic riparian mire during the spring and summer ($P<0.01$). Overall, the implications of these findings are that rheotrophic flushes and mineratrophic riparian mires (the least nutrient limited) have the greatest biogeochemical capacity to retain inorganic N and that their role at the landscape level is crucial to buffering N deposition inputs.

2. Policy Relevance

Wetlands represent ecologically and agriculturally significant components of the rural landscape often containing rare species of plants and invertebrates that support unique bird and wildlife populations. In the UK, wetlands occupy large areas of marginal and upland landscapes that are characterised by nutrient paucity and low productivity (either unmanaged, grazed or maintained for game birds). The diversity of marginal-upland wetlands, from *Sphagnum* dominated blanket peat (designated as Priority habitats by the UK Biodiversity Action Plan) to waterlogged *Juncus/Agrostis* acid grassland (many designated as EU Special Protection Areas for their provision of habitat for rare birds such as yellow wagtails), is a reflection of differences in their biogeochemistry and biodiversity e.g. ombrotrophic/blanket peatlands (Countryside Vegetation System 94-100), mineratrophic riparian peat (CVS 86-88), marshes/acid grasslands (CVS 51-55), linear riparian grasslands/marshes (CVS 41, 48). The attribution of climate and seasonality as a driver of changes wetland functions and services is clearly an important issue and will become increasingly so throughout the 21st Century (IPCC; Soil Strategy for E&W; Environment Agency 'State of Soils').

3. Objectives

In the UK, wetlands occupy large areas of marginal and upland landscapes that are often characterized by nutrient paucity and low productivity. Consequently these 'bottlenecks' of nitrogen biogeochemistry are thought to play key role in nitrogen cycling at the ecosystem, catchment and landscape scales. Nitrogen release from organic matter rich upland ecosystems typical of much of the UK is ultimately determined by the net balance between nitrogen supply and biological demand. Gross nitrogen supply is composed of agricultural and atmospheric inputs whereas biological demand is dominated by microbial immobilization and plant uptake. Excess of supply over biological demand can result in net nitrogen losses as nitrate (NO_3^-) breakthrough or in some cases as N_2O from nitrification and/or denitrification. Despite the ecological and landscape significance of these ecosystems, their role in landscape nitrogen cycling remains poorly understood. A greater understanding of the nitrogen dynamics in these wetlands is required to determine their role as regulators of nitrogen accumulation and release as breakthrough NO_3 into catchment streams. Scientific evidence is required to ; 1. to develop better landscape scale models to predict the effects of nitrogen deposition on terrestrial nitrogen dynamics and biodiversity and 2. to underpin landuse policy decision making with respect to changes EU agricultural subsidy payments to upland agri-businesses to manage landscape nitrogen dynamics.

To examine the N processes controlling nitrate leaching from soils to freshwaters in upland wetlands (provision of N data for dynamic models).

Specific hypotheses

- That seasonal variations in gross soil nitrogen fluxes would be greater than differences between wetland types.
- That nutrient poor systems would be least sensitive to seasonal variations.

4. Methods and results

4.1 Methods

4.1.1 Site and wetland characteristics

A catena of wetlands was selected at a field site in Plynlimon in Mid Wales (Figure 1). The climate is temperate/oceanic with mean annual temperature of ~9^{o}C and 2400 mm rainfall year^{-1}. The landscape is dominated by rough sheep grazed acid grassland, with ranker, peaty podzol, gley, peat and altered brown earth soils. The first of was located at the summit of a sub-catchment area at an altitude of approximately 530 meters (asl). This wetland is ombrotrophic and vegetation was dominated by *Calluna vulgaris* with some bryophytes (*Sphagnum* sp) and cotton grasses (*Eriophorum vaginatum - angustifolium*). The second wetland on the site was in a mineratrophic or riatrophic flush, with vegetation dominated by *Juncus effusus* and a mixture of grasses (*Molinia caerulea, Deshampsia flexuosa*) lying at an altitude of roughly 490 meters. The third wetland area was a riparian valley bottom system lting at the foot of the catchment next to a river at an altitude of 390 meters where vegetation consisted primarily of purple moorgrass (*Molinia caerulea*) hummocks. Climate, vegetation and soil data are shown in Table 1.

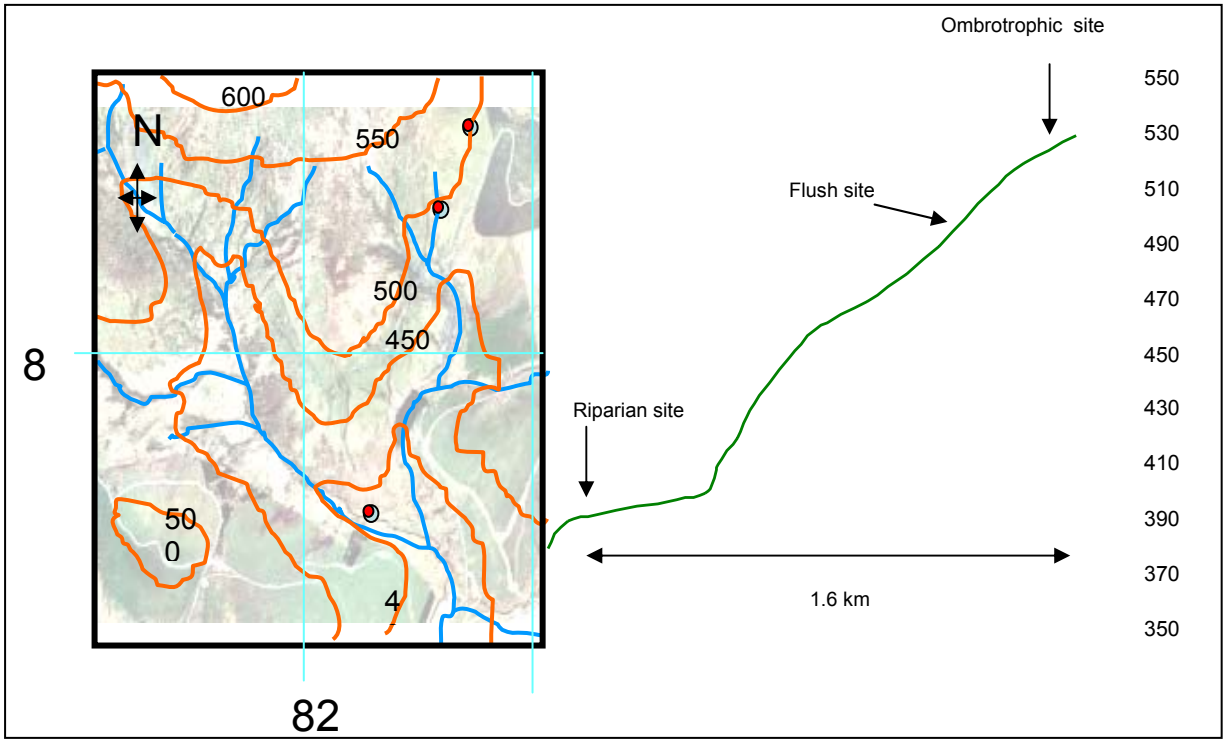


Figure 1. Location of wetland catena within the Plynlimon catchment, including side elevation.

Table 1. Plynlimon climate and wetland vegetation and soil characteristics (from Crook 2004)

	Units	**Hilltop**	**Flush**	**Valley Bottom**
Location	OS grid reference	SN825870	SN826865	SN821856
Climate				
Altitude	m asl	560	490	405
Precipitation	mm	2443	2443	2367
Air temp	ºC	8.0	9.8	9.5
Soil temp	ºC	7.5	8.3	8.3
Water table	cm below ground	-2.3	-16.4	-4.8
Soil moisture	$m^3\ m^{-3}$	0.965	0.766	0.952
Vegetation characteristics				
Biomass	$g\ m^{-2}$ (control cores)	798.07	1645.29	719.98
C:N ratio	(higher plants)	34.37	33.51	31.57
C:N ratio	(mosses*)	47.44	27.76	40.49
Description	NVC community (Rodwell, 1991)	H12	M23	M25
Soil characteristics				
Bulk density	$kg\ m^{-3}$	76.45	175.42	106.35
C:N ratio		30.73	19.39	17.18
Soil OM	g per 100 g oven-dry soil (LOI @ 500 ºC)	96.58	73.15	75.36

4.1.2 ^{15}N pool dilution

Standard methodologies enable a measure the net change in the two microbial transformations processes which affect soil inorganic nitrogen supply (ammonification + nitrification = mineralization) and microbial demand (immobilisation) (Figure 1). Using this approach is, however, problematic as changes in inorganic N concentrations and net fluxes may obscure changes or differences in the contributory gross fluxes e.g. an increase in net inorganic nitrogen supply may be due to an increase in the gross mineralization rates and / or a decline in gross immobilisation rates by the soil microbes (Schimel, 1996).

The application of ^{15}N techniques has already clarified that different ecosystem N biogeochemistry varies considerably (Hughes et al. 2005, Emmett 2007). However, there has, as yet, been no attempt to examine the relative importance of landscape and seasonality on these gross N fluxes. The assessment of the effect of these factors is all the more important considering that the primary regulators of ecosystem nitrogen transformations are biological and therefore potentially sensitive to changes in the associated abiotic conditions.

^{15}N pool dilution assays were made at all of the three wetlands on 4 separate occasions between 2005-2007 to account for seasonal variation in biological functions. At each site, four plots were marked out along a transect. Five soil cores were taken at each of these plots, two for NH_4^+ analysis, two for NO_3^- analysis and a final control core (colorimetric analysis by segmented flow autoanalyser). One of the NH_4^+ and one NO_3^- core were returned to the lab for initial prelabelling analysis. The remaining ammonium core had 15.3 atom % $^{15}NH_4^+$ added, and the NO_3^- core had 15.3 atom % $^{15}NO_3^-$ added. These were then placed back into the ground from the plot that they were excavated from and covered with soil. The control cores were also placed back in the ground (T/Umbrella method of Hughes et al. 2004). The "initial" cores were injected with either ^{15}N atom % NH_4^+ or NO_3^- in the laboratory and were then KCl extracted. The resulting solutions were then steam distilled, weighed, and then combusted using a CarloErba elemental analyser. The resultant N_2 from combustion/reduction was analysed for $^{15}N/^{14}N$ ratios using a Dennis Leigh technology IRMS (CEH Stable Isdtope Facility, Lancaster). After seven days incubation *in situ* the remaining cores were removed from the ground and the same procedure carried out. This whole process was repeated four times throughout the year to examine seasonal variations. Sampling times were April 2005, July 2005, October 2005 and January 2006. The estimates of gross mineralization, ammonification, nitrification and immobilization fluxes were made using standard ^{15}N pool dilution mass balance calculations (e.g. Kirkham and Bartholomew 1954; Murphy et al., 1999).

4.2 Results

The results of the seasonal ^{15}N pool dilution measurements on the three upland wetland systems showed that all three systems exhibited greatest NH_4^+ and NO_3^- inflow (production rates) during the summer at the peak of the growing season (Table 2).

Table 2. Gross and net NH_4^+ and NO_3^- fluxes in the wetlands derived from mass balance ^{15}N pool dilution equations; where N>INFLOW is nitrification and ammonification and N<OUTFLOW is ammonification, nitrification and microbial immobilization.

	Ombrotrophic		***Rheotrophic***		***Mineratrophic***	
			INFLOW			
2005-2006	NO_3 *nitrification*	NH_4^+ *ammonif*	NO_3 *nitrification*	NH_4^+ *ammonif*	NO_3 *nitrification*	NH_4^+ *ammonif*
Spring	2.16 ± 0.20	3.49 ±1.87	0.68 ±1.70	2.66 ±0.30	4.48 ±1.32	10.13± 4.23
summer	5.59 ±1.35	4.09 ±1.72	3.47 ±1.83	3.54 ±1.70	1.13 ±0.25	18.25 ±7.20
Autumn	3.33 ± 0.49	2.88 ±1.68	1.68 ±1.45	2.40 ±0.89	0.95 ±0.23	7.87 ±3.50
Winter	3.31 ± 2.34	0.02 ±0.08	1.45 ±1.10	0.14 ±0.09	3.21 ±1.35	0.35 ±0.16
			OUTFLOW			
	NO_3^- *immobilisation*	NH_4^+ *immobilization + nitrification*	NO_3^- *immobilisation*	NH_4^+ *immobilisation + nitrification*	NO_3^- *immobilisation*	NH_4^+ *immobilisation + nitrification*
Spring	7.18 ±0.66	5.43 ±3.67	2.37 ±2.69	3.76 ±1.20	11.09± 3.08	6.29 ±5.49
Summer	13.01 ±3.27	6.99 ±2.22	6.21 ±2.76	2.89 ±0.78	8.58± 0.77	13.48 ±3.21
Autumn	9.90 ±1.64	5.16 ±3.86	4.37 ±3.86	2.35 ±0.89	8.73± 1.29	9.74 ±4.26
Winter	7.98± 2.47	0.84± 0.16	3.52± 3.75	0.09± 0.60	12.25± 2.36	1.77± 0.64

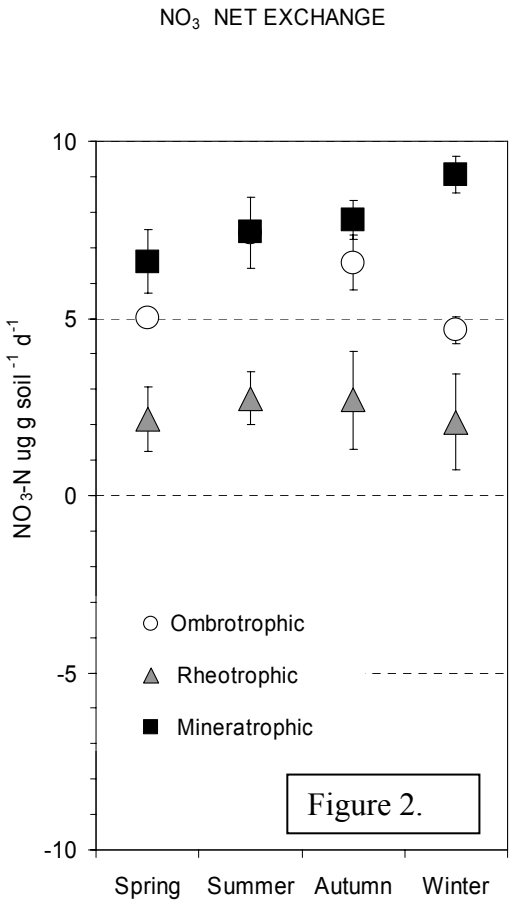


Figure 2.

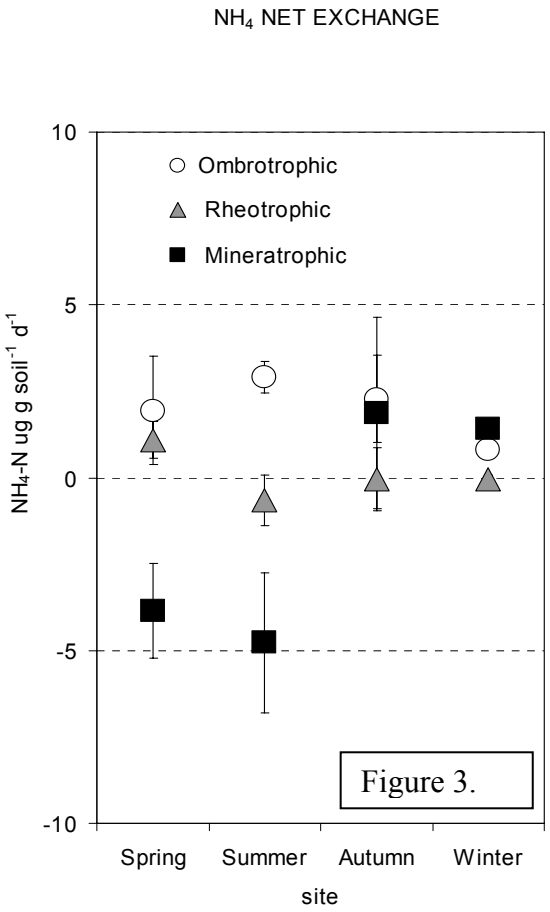
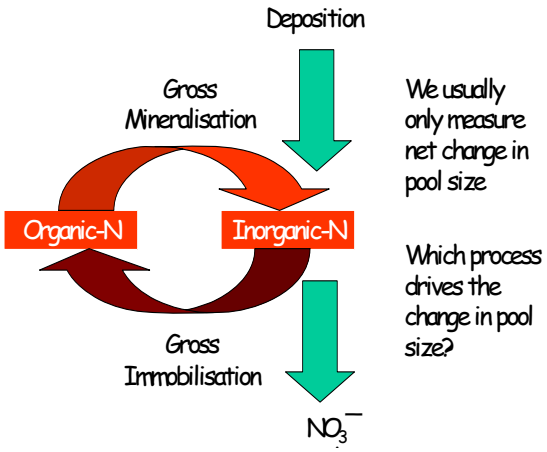


Figure 3.

The results of this work show that all of the wetlands were net NO_3^- exchangers with the rheotrophic flush being the least active (Figure 2). Season had no significant effect on net nitrate production ($P<0.01$). Despite high gross fluxes, net NH_4^+ exchange within the wetland soils was small. There was significant nitrogen retention (microbial immobilization) in the mineratrophic riparian mire during the spring and summer ($P<0.01$) (Figure 3). **There was no relationship between initial inorganic N status and the net NH_4 and NO_3 exchange fluxes of the soil.**

5. Discussion and implications

In the UK, wetlands occupy large areas of marginal and upland landscapes that are often characterized by nutrient paucity and low productivity. Consequently these 'bottlenecks' of nitrogen biogeochemistry are thought to play key role in nitrogen cycling at the ecosystem, catchment and landscape scales. Nitrogen release from organic matter rich upland ecosystems typical of much of the UK is ultimately determined by the net balance between nitrogen supply and biological demand. Gross nitrogen supply is composed of agricultural and atmospheric inputs whereas biological demand is dominated by microbial



immobilization and plant uptake. Excess of supply over biological demand can result in net nitrogen losses as nitrate (NO_3^-) breakthrough or in some cases as N_2O from nitrification and/or denitrification.

The results suggest that highest gross fluxes and nitrogen turnover were observed in the mineratrophic riparian wetland and rheotrophic flush. The mineratrophic wetland nitrate pool was significantly reduced during months of highest ammonium immobilization. The immobilisation of inorganic soil nitrogen by microbes is a potentially important determinant of the buffering capacity of wetlands for N deposition. Findings from this study support this hypothesis as highest gross fluxes were observed in the spring and summer months, suggesting that an interaction between the seasonality and the wetland type could be an important regulator of N dynamics. Overall, the implications of these findings are that rheotrophic flushes and mineratrophic riparian mires (the least nutrient limited) have the greatest biogeochemical capacity to retain inorganic N and that their role at the landscape level is crucial to buffering N deposition inputs.

Key Findings

The ^{15}N pool dilution method was developed and tested for application in water saturated organic soils and peats. Early findings from the current T-Umbrella Task 13 'Wetlands as Nitrate Regulators' show that season and climate are strong drivers of soil N biofunction. Results also indicate that there is a strong influence of wetland type on gross fluxes.

Collaborations
Initial links have been made with other CEH modeling groups (Drs C. Evans and S. Smart) particularly prospect of using landscape data for integration into upland MAGIC/SMART model runs.

6. References

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Work Package 2:

Impacts, Recovery and Processes

Task 14:
Provision and interpretation of long-term data to refine critical loads for forests, modifications required due to climate change and review of peat pH

M. Broadmeadow and E. Vanguelova

Forest Research

Task 14 - Provision and interpretation of long-term data to refine critical loads for forests, modifications required due to climate change and review of peat pH.

PIs: M. Broadmeadow and E. Vanguelova

Forest Research

1. Summary

The principal roles of Forest Research are to provide expert advice and to deliver a range of site specific environmental, soil and vegetation-related data-sets required for evaluation of the Critical Loads methodology and the further development and testing of dynamic ecosystem models including MAGIC and SAFE. The data-sets are based on measurements made in the UK plots of the EC and UNECE-ICP (Forests) Intensive Forest Health Monitoring (Level II) Network, which were established in 1994-5. An ongoing analysis of the level II data-sets provides information on the long-term dynamics of forest ecosystems and recent data have been added to the trend analysis. Key findings include the confirmation of recovery of soil solution pH from high pollution loading at the Ladybower site in the English Midlands and, in addition, recovery is becoming apparent at another acid site in Wales – Llyn Brianne. A downward trend in soil solution sulphate concentration has been observed at most sites. Foliar sulphur content at all sites has also decreased such that at Rannoch, one of the pristine Scottish sites, to a level classed as deficient. Foliar aluminium levels have continued to fall at most of the conifer sites, suggesting a general recovery from acidifying pollutant loading. The soil solution nitrogen dynamics at Tummel cannot be interpreted by dynamic models of ecosystem chemistry and, instead, a biotic explanation is offered, linked to observed peaks of litterfall together with data from the UK Forest Condition Survey. The reasons for observed trends in crown condition at the level II plots are also evaluated.

The simple mass balance equation for setting critical loads for acidity and nutrient nitrogen includes a growth uptake term for base cations and nitrogen, respectively. Updated values for this uptake term are now available based on a new set of measurements of stemwood chemistry, including ten additional sites, which were established in 2002-3. Eleven of the twenty sites were thinned during winter 2004-5 and new functions for deriving branchwood biomass have also been developed. These relationships will be used to further update the uptake term when chemical analysis of branchwood samples is complete.

The effects of predicted changes to forest productivity as a result of climate change on the base cations and nitrogen uptake term in the Critical Loads calculations are evaluated. The evaluation includes also the regional expression of the climate change impacts and effect on the Critical loads exceedance statistics. Methodology for incorporating regional variation in productivity on the growth uptake term is developed, together with an approach to represent the impacts of climate change. A summary of the approach is presented.

A review of peat pH values has been conducted with the objective of providing a robust basis for confirming or amending the critical pH for setting Critical Loads for acidity for organic soils in the UK. The review provides a basis for the re-evaluation of the critical peat pH of 4.4.

2. Policy Relevance

The critical load approach is a key element of emissions reduction policy. Its continuous evaluation and development using current data are crucial for producing updated critical loads exceedance maps for the UK and thus targeting implementation policies towards effective ecosystem recovery. For woodland, the approach must be appropriate to the breadth of environmental conditions and woodland types present in the UK, and also representative of both unmanaged woodland and current practices employed across the managed forest estate. An ongoing analysis of the level II data-sets provides an assessment of ecosystem recovery as emissions control measures are implemented. Concerns over the effects of excess nitrogen deposition and ozone pollution have come to the fore alongside climate change, which is predicted to have both direct (e.g. drought, windthrow) and indirect (pest and disease outbreaks) impacts on forest ecosystems. Some evidence is already becoming apparent. The Level II network, in this respect, provides an invaluable source of information on the condition of forests and their interaction with the wider environment. Furthermore, by validating dynamic ecosystem models using observed spatial and temporal trends in response indicators, uncertainty in the output of the models can be reduced. The models can then be used to predict the effectiveness of a range of emissions control policies on specific woodland ecosystems. A further benefit of the application of process-based dynamic models is their ability to account for predicted climate change and to distinguish between climate and pollution-driven impacts. The application of such dynamic models can predict the consequences of implementing policies aimed at climate change mitigation.

Options for climate change mitigation include an increased utilisation of biomass from existing forests for bioenergy production. Of particular interest at present to Critical Loads evaluation is the commitment made in the review of the UK Climate Change Programme (CCP06) to increase the utilisation of woodfuel from existing woodland (Defra & DTI, 2006). This commitment has been further elaborated in the Woodfuel Strategy for England (FC, 2007) produced in response to the Biomass Task Force (Anon, 2005) and set in the context of the recently published UK Biomass Strategy (Anon, 2007). The realisation of these commitments will require a step-change in harvesting activity, primarily in private-sector woodland that has been under-managed over recent decades. It is also assumed that an increased utilisation of harvesting residues and other non-merchantable fractions will contribute to meeting the target, increasing nitrogen and base cation removal at harvest, effectively increasing nutrient nitrogen and decreasing acidity critical loads. CCP06 also commits Government to 'explore the potential for a market-based greenhouse gas trading mechanism in the land management and forestry sector' – potentially raising the possibility of enhanced rates of woodland creation. Continuing critical loads development will enable an evaluation of how a range of land-use change scenarios will affect critical load exceedance, both from the perspective of reduced emissions and enhanced uptake.

3. Objectives

- Refinement of the Critical Loads methodology and assessment of critical load exceedance at the national scale for woodlands
- Evaluation of the recovery of damaged ecosystems including interactions with climate change
- Evaluation of the impact of nitrogen and acidic deposition on peat pH

4. Methods and results

4.1 Level II long-term trends

Monitoring has now been carried out across the Level II network for ten years. Large data-sets have been obtained, providing the opportunity for a closer examination of long-term trends including, for example, both the input of acidifying components in precipitation, and a range of response variables on varying time scales. Preliminary analysis suggests that for some of the data-sets, general trend analysis models cannot be applied; for example, there are often long periods of missing soil solution data in summer because samples cannot be collected. Specific models have therefore been applied to these data-sets.

Foliar data indicate that previous concerns over a risk of sulphur deficiency occurring as a results of emissions reduction, particularly in the uplands, has already been confirmed at one of the Level II sites (Rannoch), where foliar sulphur concentrations are now below the level classed as deficient (S = 0.085% dry weight, compared to the critical level of 0.1% dry weight) according to van Den Burg (1985). Foliar sulphur levels at all other sites have also decreased as a result of a steady decline in sulphate concentrations in soil solution (Figure 1). Foliar aluminium levels have also continued to fall in most of the coniferous sites suggesting general recovery from acidifying pollution loading.

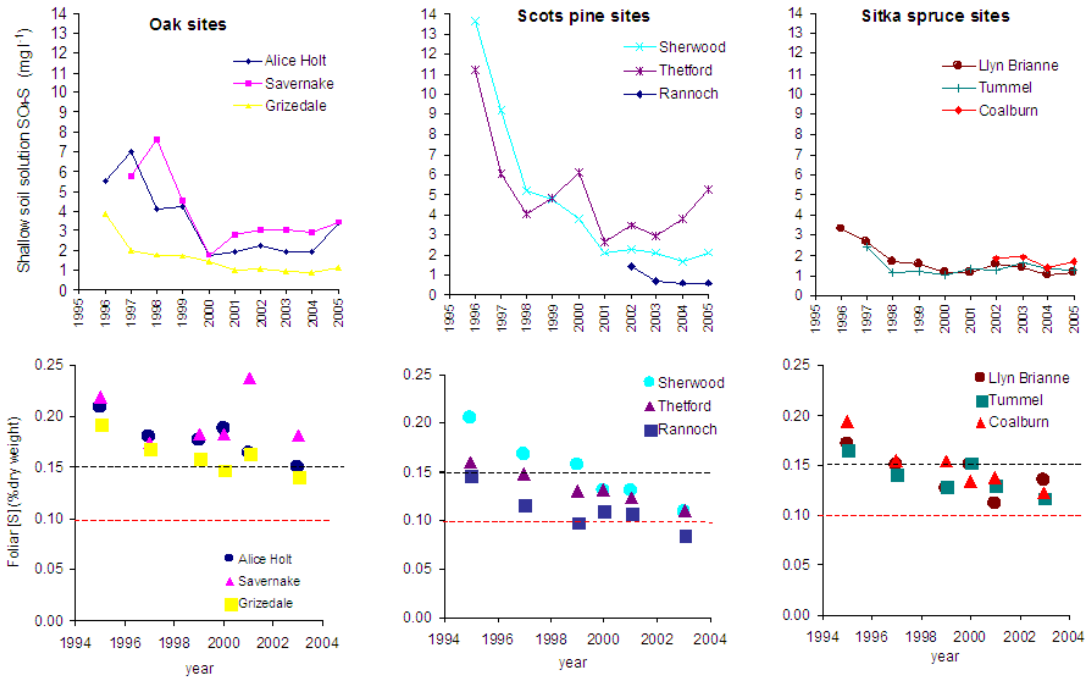

Figure 1. Soil solution SO_4-S and foliar [S] in Level II Oak, Scots pine and Sitka spruce plots between 1995 and 2005. Values shown are annual means with the red line indicating foliar [S] below which deficiency symptoms occur and the black line, optimal foliar [S] according to Van den Burg (1985).

At Sherwood, deep (50 cm) soil solution NO_3^- concentrations have declined steadily during the ten years of monitoring which resulted in a decline in tree nitrogen uptake (Figure 2). Al, Ca, Mg and K have also decreased steadily. This indicates that the rate of soil acidification resulting from high pollutant loading in the past has slowed down, with cation leaching and Al dissolution both reduced. This has been reflected in the slow but consistent recovery observed in shallow (10 cm) and deep soil solution pH for the past 10 years (Broadmeadow *et al*., 2003). A decrease in deep soil solution NO_3^- at Savernake may reflect a decline in nitrogen deposition due to a reduction in its emissions from agriculture as a result of recent changes in local agricultural practices and sources (implementation of nitrogen vulnerable zones in 1996 and 2002). This is apparent in the throughfall water collected under the oak stand for the last ten years (Vanguelova *et al*., 2007). Of the three Sitka spruce Level II plots, high nitrate concentrations in soil solution are apparent at both Coalburn and Tummel, with an increasing trend evident at Tummel. At both sites, these high concentrations are at odds with those predicted by MAGIC, assuming the low total nitrogen deposition typical of both areas. One possible explanation for Coalburn is that they may reflect mineralisation of the peat, with minimal drainage through the clay underlying the peat and effective uptake by sphagnum in the drains within the plot; nitrogen is thus not leaving the site, and this is confirmed by freshwater chemistry which does not show a significant loss of nitrogen. The high concentrations at Tummel are more difficult to explain, although the precipitation and throughfall chemistry, soil solution chemistry, litterfall and crown condition data-sets suggest that the steady increase in mean annual NO_3-N in soil solution at Tummel may be due to a biotic influence. The increase in mean annual soil solution NO_3-N for 2001, 2002 and 2003 and 2005 corresponds to two distinct peaks in NO_3-N in throughfall each year, one in early spring and one in late summer, with the same delay prior to appearing in soil solution. However, there was no corresponding increase in bulk precipitation chemistry, excluding an additional source of pollution as a possible cause. Peaks in throughfall and soil solution nitrate were followed by particularly heavy litterfall associated with attacks by the green spruce aphid, *Elatobium abietinum*. In winter and early summer the aphid feeds on mature needles which become discoloured and fall. The Forest Condition Survey results for the period 1993–2003 clearly show two occasions on which the crown density of Sitka spruce has deteriorated markedly (at a national level), which has been largely attributed to defoliation by *Elatobium* between 1996 and 1997, and between 2001 and 2002 (Hendry, 2005).

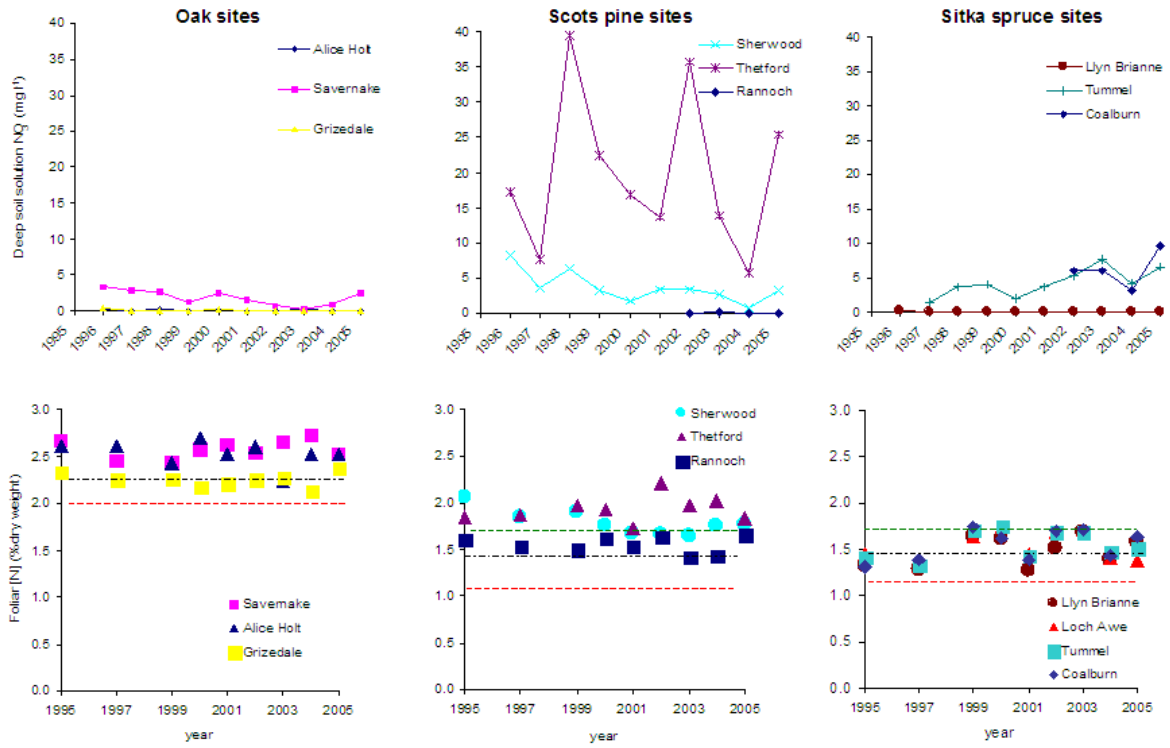

Figure 2. Soil solution NO_3^- and foliar [N] in Level II Oak, Scots pine and Sitka spruce plots between 1996 and 2004. Values shown are annual means with the red line indicating foliar [N] below which deficiency symptoms occur and the black line, optimal foliar [N], according to Taylor (1995).

Crown condition is one of the principal measures used to assess tree responses to biotic and abiotic factors at Level II sites and has also been selected by the Ministerial Conference for the Protection of Forests in Europe (MCPFE) as an indicator for sustainable forest management. Crown condition is expressed as percentage reduction in crown density relative to an 'ideal' tree.

Crown condition reflects a range of biotic and abiotic factors including insect defoliation, fungal infection, climate, air pollution and, frequently, to an interaction between these factors. A strong deterioration in mean crown density (from 14% in 1995 to 48 % in 2005) has been observed for oak at Grizedale (Figure 3a), corresponding to severe defoliation by Tortrix caterpillars, as indicated by the amount of frass and pupae numbers recorded during routine litterfall analysis. N and S deposition is relatively high compared to the other oak sites, while soil buffering is minimal due to the low soil base cation content. A similar deterioration in crown condition has been observed (from 7% in 1995 to 40% in 2005) at the Scots pine Level II site at Sherwood, situated in the previously highly polluted south Pennines (Figure 3b). Of the Sitka spruce sites, only Loch Awe showed a deterioration in crown density but the magnitude of this decline was much smaller than the oak and Scots pine sites (Figure 3c). It could therefore be inferred that high N and S deposition can contribute to reduced crown density supporting the contention of Kennedy (2002), although no causal relationship can be established.

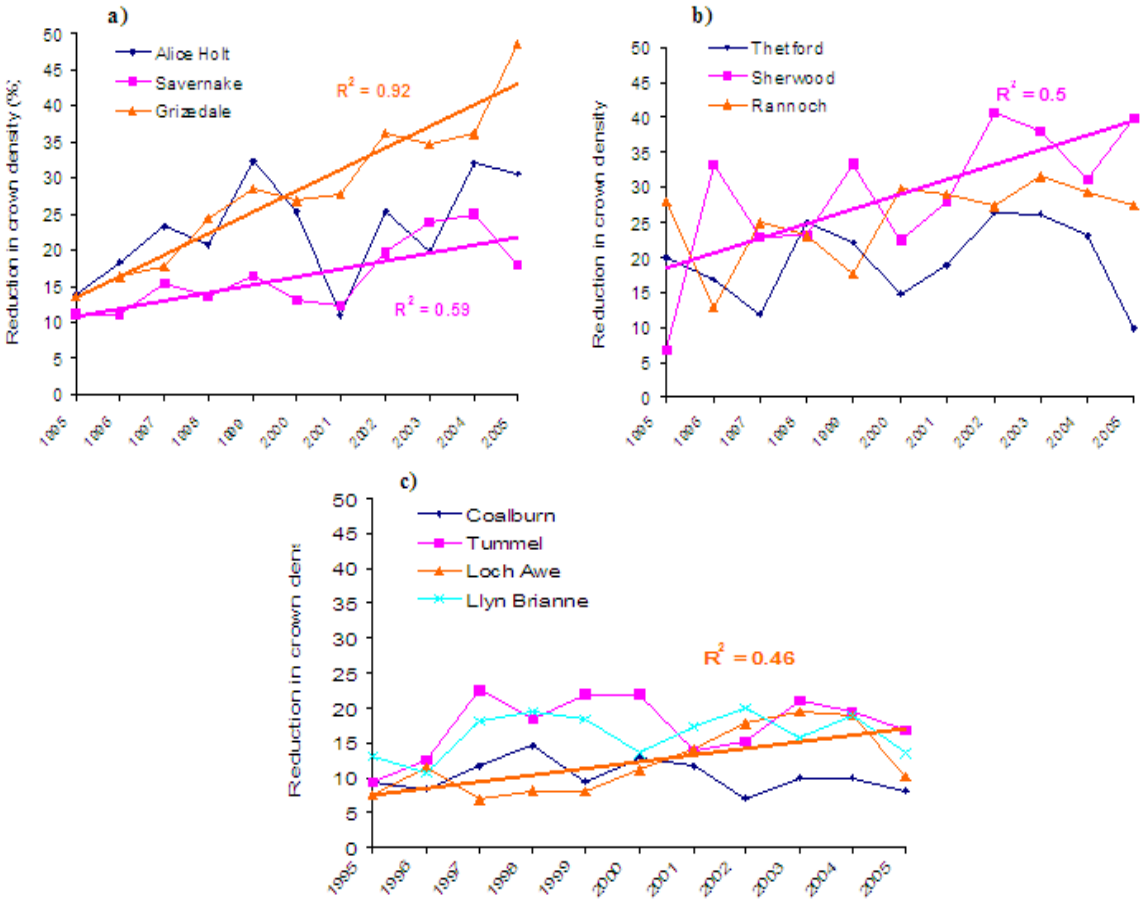

Figure 3. Reduction in crown density (%) at a) oak, b) Scots pine and c) Sitka spruce Level II intensive forest monitoring sites.

A range of insect pests are responsible for defoliation and other damage to forest trees. It is thought that many insect pests may become more damaging as a result of climate change, in part, driven by expectations that more frequent and severe summer droughts will make trees more susceptible to biotic agents. The integrated nature of monitoring across the level II network enables the relationships between climate, pollutant exposure, crown condition and insect populations, as estimated from litterfall (numbers of weevils, caterpillars and defoliating and gall-making insects) to be explored.

4.2 Critical loads – uptake term for base cations and nitrogen

The current value (2003) of the uptake term for base cations and nutrient nitrogen in the simple mass balance equation includes a contribution from branchwood for broadleaf species. The branchwood component is currently calculated assuming a fixed proportion of stemwood (18.5%) based on published allometric relationships, some of which may not be appropriate to conditions in the UK. The chemical composition of branchwood is also based on a single published value. There are concerns that uptake in branchwood is underestimated, particularly if small-diameter branchwood is taken off-site as firewood. This may become an increasingly common practice, particularly if the use of woodfuel becomes more common in small-scale power generation and CHP facilities.

It is assumed that branchwood is not taken off-site during harvesting activities in conifer stands. However, whole-tree harvesting is practiced in some regions and on some sites. Various options are available for whole tree harvesting of conifer species, and the implications of each of these for critical load exceedance and long-term site sustainability will differ. In addition, an improved knowledge of the contribution that branchwood may make to the growth uptake term would improve the knowledge-base on which guidance to manage nitrogen sensitive woodland is based, particularly for heathland sites.

The updating of the growth uptake term, with an emphasis on the contribution from branchwood, has been approached through two separate sampling strategies. Improved estimates of stemwood chemical composition have been obtained by coring five sample trees at each of the twenty Level II plots during February and March 2004 (see Broadmeadow *et al*., 2004). These data have been used to update the growth uptake term (Table 1). Branchwood chemical composition is based on values for 'outer' wood; specific branchwood values will be measured on samples collected during routine thinning operations carried out in the Level II sites.

Table 1. Updated growth uptake term of the simple mass balance equation for acidity and nutrient nitrogen critical loads calculation. Values are expressed as kg ha^{-1} yr^{-1} except for total base cations (BC), which is expressed as keq ha^{-1} yr^{-1}. Average productivity class (GYC – general yield class: m^3 ha^{-1} yr^{-1}) is also given.

	Annual growth uptake (kg ha^{-1} yr^{-1})							
	2003 analysis		Original sites		New sites		All sites	
	Bl	Con	Bl	Con	Bl	Con	Bl	Con
GYC (m^3 ha^{-1} yr^{-1})	-	-	5.0	15.9	6.3	15.3	5.9	15.7
N	5.86	2.94	3.20	2.24	2.55	2.49	2.71	2.31
K	3.05	2.27	1.69	1.79	1.95	2.14	1.88	1.89
Ca	5.28	3.22	4.76	3.23	2.72	5.81	3.2	4.0
Mg	0.57	0.58	0.25	0.54	0.54	0.65	0.46	0.57
BC (keq ha^{-1} yr^{-1})	0.23	0.16	0.17	0.15	0.14	0.23	0.15	0.17

Eleven of the twenty Level II plots were thinned in early 2005 as part of the routine stand management cycle. Branch biomass assessments were made on ten selected sample trees across the full diameter range in each plot. Conventional assessments of stem volume were carried out, plus an additional assessment of sawlog (>16 cm diameter) volume. Trees were then separated into five components; stemwood, 'measurable branchwood' (>7 cm diameter), brash (<7 cm diameter), non merchantable stemwood (stump and saddle) and deadwood. Each component was weighed separately, with sub-samples of each chipped and removed for moisture content determination and chemical analysis. Samples for chemical analysis were dried at 70°C and stored prior to analytical preparation. Moisture content was calculated as weight loss after drying at 105°C for 48 hours. For the conifer plots, foliage was included as part of the brash fraction.

Preliminary analysis indicates that for some species, the use of published allometric models, in many cases based on non-UK data, may introduce significant errors in estimates of aboveground biomass and thus nutrient nitrogen and base cation uptake. The error introduced by assuming the default biomass expansion factor (BEF) of 1.18 was largest for beech. The other noticeable deviation from the default BEF was observed in a young Norway spruce plot (Kiddens), where a very high BEF was derived. This is not surprising since it is widely known that stemwood comprises a smaller proportion of total biomass in young trees; however, this would have serious implications for critical load modelling and assessment of

long-term site sustainability if short rotation forestry for bioenergy production expanded significantly. The study also indicates that simple biomass expansion factors may be inappropriate for young trees, and that diameter or height related functions are more appropriate.

4.3 Interactive effect of climate change on critical loads

The uptake term for woodland for base cations and nutrient nitrogen within the critical load equation is based on assumed growth rates of the ten Level II plots. In turn, the yield models from which the assumed growth rates are calculated are empirical, based on historic mensuration data and unable to account for the predicted effects of climate change. Uptake rates cannot be adjusted using a single correction factor as a result of regional variations in the effects of climate change that are predicted; growth rates are likely to increase in most upland areas as a result of climatic warming and rising atmospheric carbon dioxide levels. In contrast, growth rates in the lowlands, particularly in southern England are predicted to fall significantly as a result of the predicted increase in frequency and severity of summer droughts.

Regional variation in species distribution has been introduced using data provided by the National Inventory of Woodland and Trees (NIWT), broken down by FC Conservancy. Regional variation in growth rate, under both current and future climate scenarios (UKCIP02: Hulme *et al*., 2002), has been introduced into the Critical Loads methodology using the Ecological Site Classification decision support system (ESC: Pyatt *et al*., 2001; Broadmeadow *et al*., 2005). The approach adopted is as follows: 1) Major species groupings, as given in the summary tables in the regional NIWT Inventory reports, are incorporated as gridded layers, with a single % value given for each grouping for all 5km grid-squares in each conservancy (Table 2); 3) For each species, a nutrient uptake rate per unit yield class is calculated, based on Level II wood nutrient (see section 4.2) content data and FC Booklets 39 and 48, as outlined in Hall *et al*., (2003); 4) ESC grids are provided for all species groupings and all UKCIP02 scenarios; 5) Uptake is calculated for managed broadleaf and conifer woodland separately, which is not based on all species present, but on those defined within the major species groupings; 6) The new uptake values will be used to calculate the Critical loads exceedances.

Table 2**.** Forest area broken down by species groupings based on data from the National Inventory of woodland and trees (NIWT: FC, 2001-2003)**.**

Region	**Percentage of each species grouping**					
	Pine	**Sitka spruce**	**Spruce larch, fir**	**Oak**	**Beech**	**Ash, birch sycamore**
East Midlands	18.0	0.4	6.6	20.5	4.1	34.5
East England	27.0	0.0	4.5	18.4	4.7	24.0
London	1.3	0.0	0.2	32.1	6.7	30.6
Northeast England	15.1	46.4	11.4	5.4	3.1	13.3
Northwest England	9.2	23.7	12.0	14.3	4.4	23.3
Southeast England	13.6	0.1	6.8	20.3	10.8	26.6
Southwest England	6.5	3.3	16.3	13.4	7.2	15.8
West Midlands	11.5	2.4	15.6	21.8	4.3	25.2
Yorkshire & Humberside	17.2	10.6	14.3	12.5	7.2	27.4
Highlands and Islands	46.9	27.4	6.8	1.8	0.2	11.5
Grampian	42.4	25.9	14.5	0.4	0.8	8.0
South Scotland	8.8	61.9	10.4	2.2	1.1	6.9
North Wales	6.7	36.4	17.4	16.3	3.1	12.1
South Wales	5.5	33.9	20.0	16.3	3.2	17.2

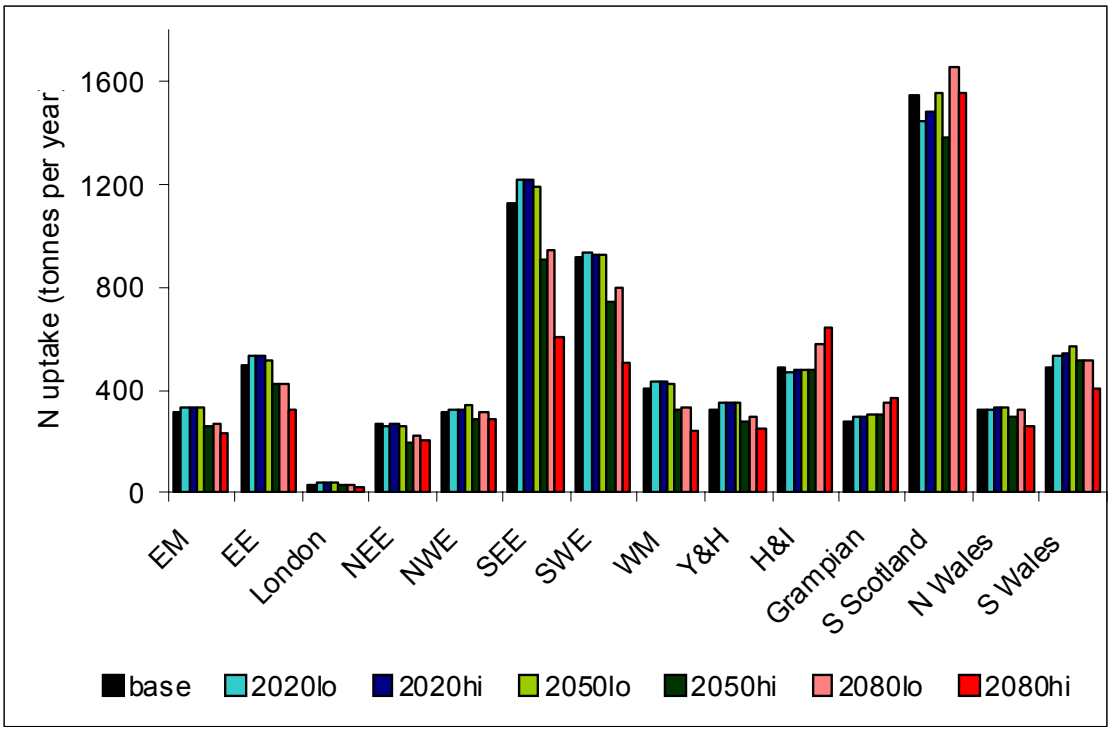

Figure 4. Predictions of the impact of climate change on total nutrient N uptake broken down by Forestry Commission Conservancy. South Scotland is an amalgamation of Perth and Argyll, Central and South Scotland Conservancies.

Regional variation in the impacts of climate change is clear (Figure 4), although this picture is complicated by the direction of change being dependent on both timeframe and emissions scenario. Initially, growth and therefore uptake are predicted to increase as a result of climate change in most regions and, up to the 2050s Low emissions scenario, impacts are relatively minor. However, towards the end of the century growth uptake is predicted to decline significantly in the southern half of England, contrasting with an increase across Scotland. Growth uptake in Wales is predicted to be relatively unaffected. Nationally, total N deposition is predicted to rise by 4% to a maximum under the 2050s Low emissions scenario and then to fall to a level 19% below the baseline under the 2080s High emissions scenario. Impacts on Critical Load Exceedance statistics will be made available elsewhere (Broadmeadow and Hall, 2007). It is important to note that this analysis does not account for the beneficial effects of rising CO_2 levels on tree growth, the impacts of extreme events including pest and disease outbreaks or climate change-driven changes in species choice or forestry policy, as alluded to in England's revised Forestry Strategy (Defra, 2007).

4.4 Pristine peat pH review

Supporting data for setting the critical pH of peat soils in the UK is limited, and based upon a single experiment using Calluna peat. The current value of 4.4 is high in comparison to that proposed in the UNECE mapping manual, and also high compared to the majority of forest soils on which there is no evidence of pH-related damage. This desk-study identifies sources and collate data for 'pristine peats' in the UK including information on the vegetation from which they were derived. These data are compared with non-UK data-sets (e.g. Scandinavia) where similar peats exist, providing a robust analysis of peat pH in functioning peatland ecosystems and a basis for an informed change to the critical peat pH adopted by the UK.

A picture of a "pristine" peat pH in the UK was built by studying the memoirs and interpreting the information and data from the Scottish soil survey carried out from the pre-war years to the late 1980s. Figure 5 shows the pH of the surface horizon of ombrotrophic peat bogs. These have been subdivided into disturbed bogs (blue), unknown bogs (red) and pristine bogs (yellow) according to site history. The average pH of each of these groups are 3.6, 3.9 and 3.6 respectively. Of the peats that were described, the majority were dominated by *Sphagnum*, the rest were a mix of *Sphagnum, Eriophorum* and *Calluna*. The *Sphagnum* peats had an average pH of 3.6. It was not possible to distinguish pH differences as a result of vegetation type. A comparison of the location of the bogs, their pH and the acidic deposition in that area suggest that acidic deposition is not a major factor leading to significant acidification. However, a study of 123 peatland sites (most probably not pristine) in Scotland (Skiba *et al.*, 1989), reported that the most acidic peats (pH($CaCl_2$)≤3.0) and with lowest base saturation (≤10%) are primarily found where acid deposition was greater than 0.8 kg H^+ ha^{-1} yr^{-1}.

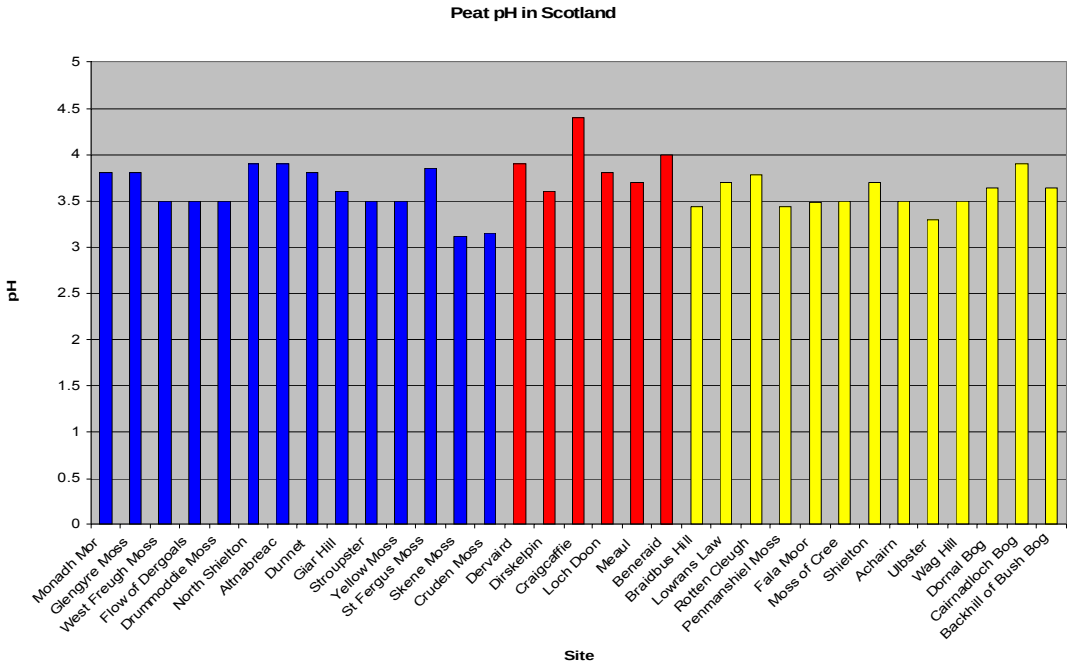

Figure 5. Peat pH in Scotland. Analysis based on data from Soil Survey of Scotland Memoirs.

The National Soils Inventory in England and Wales in 1980 included 1281 points on non-agricultural land. The re-survey in 2003 (National Soils Resources Institute) revisited 580 of these points. Grouped by major soil group, 72 of those were classified as peat soils. If an average is taken for all locations, a small but statistically insignificant increase in pH from 3.9 to 4.0 was apparent between 1980 and 2003. If grouped by land-use, 34 of the 580 were classified as bogs, with an average pH of 4.3 in 1980. This value was unchanged in 2003.

In Finland during 1975-2000, an extensive study covering about one million locations was conducted to assesses the peat reserves of Finland together with their potential for exploitation. Figure 6 shows the average pH through the profile of peatland across Finland. Peats are more acid in the south and west which coincide with the more pristine areas, however the differences are better explained by differences in the underlying geology and vegetation. In the north there is much more calcium and magnesium in the bedrock and

glacial till, which is the main soil type, than in south and central Finland. In south and central Finland Sphagnum bogs predominate (S.fuscum, S. angustifolium, S. magellanicum, S. ledum) which have a naturally low pH. In northern Finland, the dominant vegetation and cover type is fens with herbs. In central Lapland this consists mainly of plants such as *Bryum*, *Scorpidium*, *Drepanocladus*, *Sphagnum warnstorfii*, and variuos species of sedge (*Carex*).

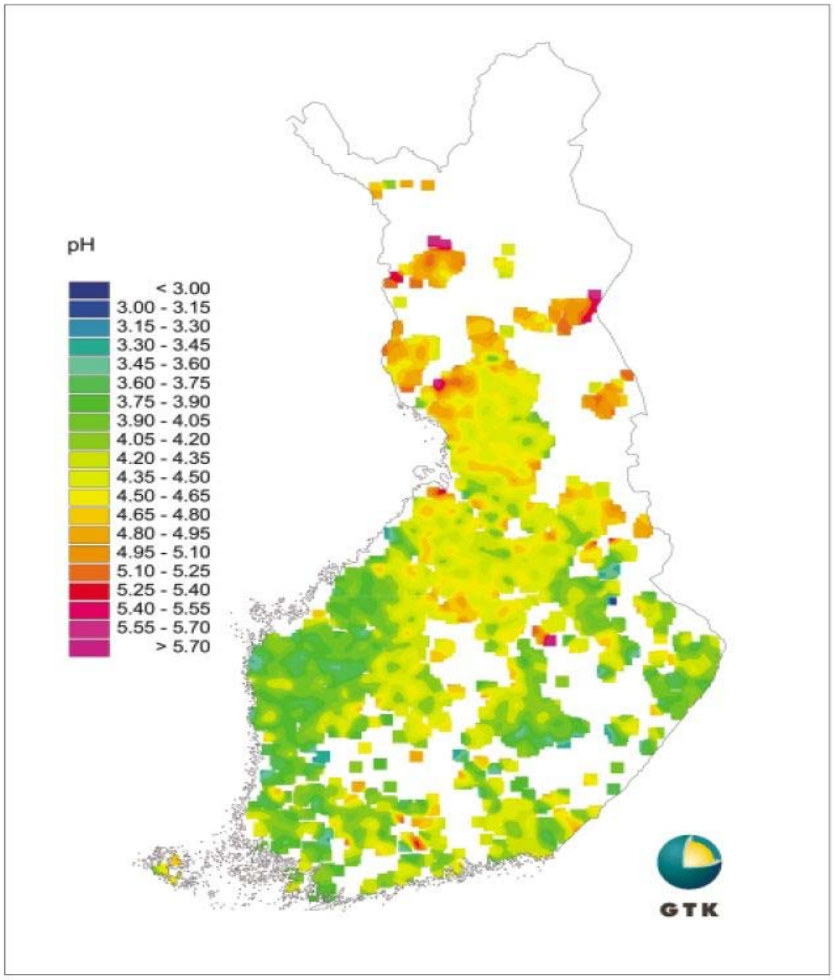

Figure 6. Regional variation of peat pH in Finland (white area on the map – no research data in GSF's files).

Peatlands acidify naturally as they develop. The natural pH of a peatland depends on what stage it is at in its succession towards a rain fed ombrotrophic bog. Expected vegetation at different values of pHs and fertility are summarised in Williams (2006). Reviewing data from the UK and Scandinavia has shown peat bogs surviving at pHs below 4.4 suggesting the acid deposition load that would give rise to an effective rain pH value of 4.4 is too high. The natural pH of fens and bogs, or rheotrophic and ombrotrophic mires differs. It therefore seems sensible to have separate critical pHs for different peat types and that a single critical value of 4.4 is too simplistic.

5. Discussion and implications

The key deliverables of Forest Research in this contract were:

- Provision of expert advice on woodland ecosystems;
- Provision of compatible, quality controlled data-sets of driving, state and response variables for Level II sites
- Revision of information on peat pH in order to inform the need for reevaluation of its critical pH value for setting Critical loads for acidity for organic soils.
- Provision methodology for the evaluation of the climate change driven impacts on critical loads for nitrogen and acidity through changes in the uptake term for nitrogen and base cations.

The analysis presented here demonstrates the value of the continuing development and evolution of the Critical Loads methodology. In particular, by basing monitoring and methodological development on the same sites, coherent data-sets can be brought together. This has proved particularly important for the development of the growth uptake term, allowing future changes in forestry and renewable energy policy together with the predicted effects of climate change to be evaluated.

Data from the Level II network have demonstrated continuing recovery from high levels of acid deposition in the past on some sites. However, on other sites, apparent trends in nitrogen dynamics are explained by biotic factors rather than as a result of a changing deposition climate. This evaluation has only been possible because of the range of measurements made on site, again demonstrating the value of an integrated monitoring network.

The preliminary analysis of climate change impacts on critical loads exceedance acting through the growth uptake term indicate that at a national level, exceedance may not be greatly affected. However, regional impacts will vary greatly and, in the case of nutrient nitrogen, the largest reduction in the growth uptake term will occur in Southeast England – the region with the highest deposition and level of exceedance. Changes in forestry practice, driven by renewable energy strategies, may counter these predicted changes through increasing removals. Indeed, it is important to note that over the next 30-50 years, it is likely that that changes to forestry and energy policy and practice are likely to have far greater impact that than climate change. Ongoing Critical Load development and evaluation will allow the impacts of potential changes in policy to be further evaluated.

Publications arising from this and associated work since the beginning of this contract are as follows:

Broadmeadow, M. 2004**.** Woodland and our changing environment. *In Brief Fact sheet,* Forestry Commission, Edinburgh.

Broadmeadow, M., Benham, S., Wilkinson, M. 2004. Growth uptake of heavy metals by British Forests: implications for critical load mapping. Report under Defra contract EPG 1/3/188 - further development of an effects based approach for cadmium, copper, lead and zinc**.** Forest Research, Farnham**.**

Durrant, D., Moffat A.J. and Barsoum, N. 2005**.** Annual progress report on intensive forest monitoring activities in the UK. Annual report to EC and UNECE-ICP (Forests). Forest Research, Farnham.

Langan, S.J., Hall, J., Reynolds, B., Broadmeadow, M., Hornung, M., and Cresser, M.S. 2004**.** The development of an approach to assess critical loads of acidity for woodland habitats in Great Britain. *Hydrology and Earth System Science* 8**,** 355–365.

Langan, S.J., Martinson, L. and Vanguelova, E. 2005**.** Dynamic modelling of UK forest soils to changes in acid deposition using the SAFE model. *The Science of the Total Environment* (to be submitted).

Pitman, R. and Vanguelova, E. 2006**.** Influences of stand age and soil propoerties on forest biodiversity. Environmnetal and Human Sciences Research Highlight 1. In: Forest Research Annual Report and Accounts 2004-5. The Stationery Office, Norwich.

Pitman, R.M., Poole, J. and Benham, S. 2007**.** A comparison of Ellenberg values from the ground flora in a chronosequence of lowland oak and Corsican pine stands with forest soil properties. Submitted to *Journal of Ecology*.

Vanguelova, E.I., Barsoum, N., Benham, S., Broadmeadow, M., Moffat, A., Nisbet, T. and Pitman, R. 2007**.** Ten years of intensive environmental monitoring of British forests. FC Information note (In press).

Vanguelova, E.I., Nortcliff, S., Moffat,A., Kennedy, F. 2005**.** Morphology, biomass and chemistry of fine roots of Scots pine (*Pinus sylvestris*). *Plant and Soil* 270, 233-247.

Vanguelova, E., Moffat, A.J. and Lynch, J**.** 2006. Impact of forestry on soil quality in the UK. Expert Workshop on Definition of Best Indicators for Biodiversity and Soil Quality for Life Cycle Assessment (LCA), 12-13 June, University of Surrey.

Vanguelova, E.I., Nortcliff, S., Moffat,A., Kennedy, F. 2007**.** Short term effects of acid manipulation on soil, soil solution and fine roots of Scots pine (*Pinus sylvestris*) stand on a podzol. Plant and Soil (In press).

6. Possible future work

Provision of data arising from the routine activities of the Intensive Forest Health Monitoring (Level II) Network, that are required for steady state mass balance or dynamic modelling of critical loads of nutrient nitrogen or acidity and their exceedance and/or recovery. An annual update to the data-sets will be provided together with a statistical analysis and interpretation of the results in terms of changes to the pollution climate and the strength of any apparent recovery from historic pollution or further acidification or eutrophication.

The tree species that are planted at restocking may not be the same as are currently present as a result of changing forestry policy and adaptation measures designed to address the threats of climate change. Furthermore, growth rates of woodland are also likely to change as a result on the progressive impacts of rising atmospheric carbon dioxide levels and climate change. Changes in both species and growth rates will also affect the base cation and nutrient N

uptake term in the critical loads calculation and have consequent effects on exceedance statistics. We propose to carry out a case study to identify the implications for nutrient N and acidity critical loads of 1) changing species choice at re-stocking to maximise resilience to climate change and 2) changing species choice at re-stocking to reflect current nature conservation and forestry policies. The objectives will be met by carrying out a GIS case study of a defined forest area covering a range of base rich (e.g. at Alice Holt) and base poor soil types (e.g. Ladybower) in an area of high exceedance. Existing critical loads methodology using the simple balance approach will be applied using measured nutrient concentrations in stemwood, bark and branchwood from the Level II network together with literature values as appropriate. Current tree species distribution will be provided from the FC sub-compartment data-base, with information on current and future yield class (using the UKCIP02 climate change scenarios) provided by the Forestry Commission's Ecological Site Classification decision support system. A detailed characterisation of the deposition climate will be given by the Level II deposition data-set from Alice Holt and Ladybower, together with ancilliary information on mineralisation and leaching. The results from this case study will be used as a base for future evaluations of how climate change policy and adaptation measures affect the Critical loads for forestry at national level.

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Work Package 2:
Impacts, Recovery and Processes

Task 15:
Analysis of monitoring data

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Task 15 - Analysis of monitoring data

PIs: S. Smart, L. Maskell and A. Scott

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1. Summary

Understanding the increasing abundance of nutrient-demanding species in upland infertile habitats.

- Plant species less typical of upland vegetation types and more typical of lowland semi-improved grasslands, have increased in occupancy in Countryside Survey (CS) plots in upland Britain between 1978 and 1998. We sought to explain this pattern by correlating spatial and temporal changes in abundance of this species group with potential drivers including change in growing season length, change in sheep density between 1969 and 2000, sheep numbers in 2000, cover of intensive Broad Habitat in 1998 and total modeled N deposition in 1996 additionally broken down into NHx and NOy as separate explanatory variables.
- When variation due to other drivers was covaried out, the probability of at least one nutrient-demanding, lowland mesophyte being present in an upland grassland, heath or bog plot in Countryside Survey data in 1998 was best explained by long-term average annual temperature and total modeled N deposition in 1996.
- However, the richness of nutrient-demanding, lowland mesophytes in CS plots in 1998 was positively correlated with sheep grazing intensity and cover of intensive Broad Habitat in each 1km square.
- No significant correlations were detected between change in richness or presence of nutrient-demanding species over time (1978 to 1998 or 1990 to 1998).
- In conclusion, a warmer, more lowland climate plus high N deposition seem to track the incursion of these nutrient-demanding species into upland Britain but more intense agricultural activity, particularly sheep grazing, is best correlated with a greater richness of these species per square metre, and therefore with the most marked changes in local species composition and vegetation character.

Deriving botanical indicators of N deposition across upland, infertile vegetation types in Britain

- Previous work has shown that the large-scale eutrophication signal across Britain can be attributed to modelled N deposition at the level of individual plant species. This relies on correlating species changes with N deposition.
- Hitherto, signal attribution has not taken into account the competing effects of other human impacts including climate change and land-use.
- We sought to identify individual plant species as indicators of the unique signal of N deposition having covaried out relationships with these other drivers of ecological change.
- Despite the fact that the scale at which driving variables are resolved was much bigger than CS sampling plots, modeled NHx deposition displayed significant correlations

with 41% of the common higher plant species present in upland CS plots in the three survey years (total of 146 plant species tested).

- A smaller subset carried the strongest unique signal of NHx deposition.
- Overall more species-level differences in abundance between CS squares could be attributed to spatial differences in NHx deposition than to change in sheep density or growing season length.
- Finally, these results must be strongly qualified by the statement that they are correlative patterns that could be concealing other causal links, rather than causal relationships inferred more strongly from experimental manipulations.
- Further work is required to establish the potential involvement of reduced SOy deposition in driving the observed eutrophication response.

2. Objectives

Agreed Milestones (M) and Deliverables (D)

M1.1	*Analysis of temporal and spatial change in lowland nutrient-demanding species*	*Dec 2005*	**Complete**
D1.1	*Report describing relationship between botanical change in GB uplands and driving variables including N and S deposition.*	*Mar 2006*	**Partly complete**
M1.2	*Completion of indicator species analysis*	*Mar 2006*	**Complete**
D1.2	*Report presenting indicator species and relationships to driving variables.*	*Dec 2006*	**Complete**

3. Introduction

The most important drivers of vegetation change, in terms of their prevalence across upland Britain between 1978 and 1998, were increases in grazing pressure and nitrogen deposition (Smart et al 2005). Most recently, climate change has also increased in importance as a potential driver. During this period, clear signals of eutrophication have been detected in upland vegetation types involving a gradual increase in abundance of a small number of nutrient-demanding mesophytes more typical of semi-improved, lowland grasslands (Smart et al 2005, Smart et al 2003a,b). Given the operation of multiple drivers during the interval, a key question is to what extent observed species compositional changes uniquely reflect N deposition. This report takes two approaches to advance our understanding of the relative contribution of key human impacts in driving change in upland infertile vegetation types. In the first section, the joint abundance of a specific set of species in upland Countryside Survey (CS) plots was used to create three response variables; the proportion of plots occupied per 1km square in 1998, change in proportion of plots occupied over time, and species richness of the group per plot in 1998. The group of species used comprise those known to have increased in abundance across upland heath, bog and grassland between 1978 and 1998. These are listed in table 1.

Table 1. List of atypical species found to have increased in infertile upland habitats (heath, bog and upland grasslands) between 1978 and 1998 (see Smart et al 2005 and www.cs2000.org.uk/M01_tables/reports).

Agrostis stolonifera	*Urtica dioica*
Cardamine hirsuta/flexuosa	*Poa trivilais*
Cerastium fontanum	*Poa annua*
Cirsium arvense	*Arrhenatherum elatius*
Cirsium vulgare	*Plantago major*
Cynosurus cristatus	*Anthoxanthum odoratum*
Deschampsia cespitosa	*Ranunculus repens*
Festuca rubra agg.	*Poa trivilais/nemoralis*
Holcus lanatus	
Holcus mollis	
Lolium perenne	
Ranunculus ficaria	
Rumex acetosa	
Stellaria uliginosa	

The second section also focuses on upland habitats in GB but examines the signals of all species frequent enough to have shown significant change over the 1978 to 1998 period. The objective here was to search for botanical indicators that best reflected the unique impact of N deposition (both positive and negative) in the presence of other potential drivers.

3.1 Understanding the increasing abundance of nutrient-demanding species in upland infertile habitats.

3.1.1 Methods

The 1998 Countryside Survey data set was used to determine the spatial variation of atypical species in relation to a number of explanatory variables. The 1998 survey data set contains the largest number of plots (16851) of different sizes and types. The number of plots containing at least one atypical species in a square was counted and the proportion of such plots in a CS 1 km square calculated. Plots were only selected from upland environmental zones 3,5 and 6 and in the upland vegetation types heath/bog and moorland grass/mosaics (Bunce et al 1999; Smart et al 2005). In addition, only areal and linear plot types most likely to be affected by grazing animals in the unenclosed uplands were included. Hence field boundary and road verge plots were excluded. The mean richness of atypical species per plot per square was also calculated and used as an additional response variable.

Generalized linear models (proc genmod, SAS Insitute) were used to determine which of a range of potential driving variables were best correlated with the proportion of plots in each CS 1km square containing at least one member of the species group (Table 1). Two types of analysis were carried out: First, tests of the explanatory power of each driver were carried out separately for each driver. However because drivers can operate in a correlated fashion across GB, each driver was additionally tested by entering it last into a sequential model after all

other variables (Type 1 tests). These analyses were then repeated using mean richness of nutrient-demanding species per plot per square. Explanatory variables are listed in Table 2. Change in the proportion of nutrient-demanding species over time in relation to environmental variables was also analysed using the above methods between 1990 and 1998, and between 1978 and 1998.

Table 2. Explanatory variables used to detect partial and unconditional correlations between botanical change in upland GB and gradients of the intensity of key human impacts.

Driver	Spatial scale	Temporal scale	Source
Sheep density	2km sqr	2000	MAFF census[1]
Change in sheep density	2km sqr	1969-2000	MAFF census[1]
Mean min Jan temp	5km sqr	1961-1999	UKCIP[2]
Mean max July temp	5km sqr	1961-1999	UKCIP[2]
Change in annual growing season length	5km sqr	1961-1999	UKCIP[2]
Atmospheric N deposition	5km sqr	1996	Models and measurements[3]
Intensive Broad Habitat cover	1km sqr	1998	Countryside Survey field maps[4]

3.1.2 Results

Spatial analysis

When tested individually, therefore with no other covariates present, all explanatory variables were significantly correlated with proportion of plots occupied by nutrient-demanding species. However, this could be because drivers were correlated in their operation across the sampling domain and so shared explanatory power because their signals overlapped. We know, for example, that both increased sheep grazing and heightened N deposition can favour nutrient demanding grasses but in CS surveillance data these 'treatment' effects could well be spatially confounded (termed multi-collinearity). Hence, an apparent signal due to N deposition could be due to the fact that the N deposition gradient is strongly correlated with the sheep grazing intensity gradient and vice versa. The type 1 tests quantify the partial explanatory power of each driver and therefore exclude any overlapping variance beween drivers. The discrepancy between the two sets of results gives an indication of the extent of this spatial overlap among drivers. When the type 1 test results are examined (Table 3) the number of significant partial predictors drops dramatically. NOy rather than NHx now emerges as the only unique N deposition predictor part from Total Nitrogen. This probably

[1] Downloaded for each 5km sqr containing each CS 1km sqr from the EDINA AgCensus database at www.edina.ac.uk.
[2] Long term annual average for the 5km sqr containing each CS 1km sqr. Downloaded from www.met-office.gov.uk/research/hadleycentre/obsdata/ukcip/index.html.
[3] See NEGTAP (2001) available at www.nbu.ac.uk/negtap/finalreport.htm.
[4] Sum of the percentage of improved grassland+arable+urban/built in each 1km sqr.

reflects the strong correlation between change in sheep grazing intensity and NHx deposition but low correlation between change in sheep grazing and NOy deposition. It does however, indicate that NOy appears to have an additional and independent role as a correlated driver of change in the proportion of atypical species in upland habitats across Britain.

Table 3. Results from analysis of plots from aggregate classes 7 and 8, environmental zone 3,5,6 (uplands), area plots (X,Y,U, SW) with proportion of plots in a square containing an atypical species as the response variable and climatic, modeled N deposition and grazing variables as explanatory. Both tests incorporating all variables (type 1 tests) and individual tests are included.

	Type 1 tests		**Direction of correlation**	**Individual tests**		**Direction of correlation**
	Chi sqr	p		chisqr	p	
Mean min jan	2.83	0.09		0.73	0.4	
Total Nitrogen	4.08	**< 0.05**	+	51.82	**<0.001**	+
NHx	0.15	0.69		46.42	**<0.001**	+
NOy	5.5	**< 0.05**	+	39.87	**<0.001**	+
Change in sheep numbers	0.6	0.43		17.73	**<0.001**	+
Total sheep 2000	0.14	0.71		12	**<0.001**	+
Growing season change	1.35	0.25		14.09	**<0.001**	-
Average temperature	4.8	**< 0.05**	+	42.86	**<0.001**	+
Amount of intensive Broad Habitat in square	0.71	0.39		5.13	**<0.05**	+

When mean species richness per plot per square in 1998 was used as a response variable instead of the proportion of plots occupied, the results were rather different (Table 4). Fewer explanatory variables showed significant relationships. Interestingly, agricultural disturbance in the form of either cover of intensive Broad Habitat per square or sheep density, were associated with a higher richness per plot as expected given the habitat affinities of the species group.

Table 4. Results from analysis of plots from aggregate classes 7 and 8, environmental zone 3,5,6 (uplands), area and stream plots (X,Y,U, SW) with mean richness of atypical species per square as the response variable and climatic, N deposition and grazing variables as explanatory. Both tests incorporating all variables (type 1 tests) and individual tests are included.

	Type 1 tests		**Direction of correlation**	**Individual tests**		**Direction of correlation**
	Chi sqr	p		chisqr	p	
Mean min jan	3.32	0.07		0.01	0.94	
Total Nitrogen	0.98	0.32		0.12	0.72	
NHx	3.57	0.06		1.34	0.24	
NOy	1.55	0.21		4.33	**<0.05**	-
Change in sheep numbers	0.02	0.89		5.75	**<0.05**	+
Total sheep 2000	0.07	0.79		4.89	**<0.05**	+
Growing season change	0.16	0.69		0.01	0.93	
Average temperature	8.41	0.48		0.45	0.48	
Amount of intensive Broad Habitat in square	4.54	**<0.05**	+	15.69	**<0.001**	+

Temporal analysis

There were no significant relationships between change in the proportion of plots containing an atypical species versus nitrogen deposition, climatic and grazing variables for either the time period 1978 to 1998 or 1990 and 1998 in upland habitats.

3.1.3 Discussion and implications

Strong correlations were detected between spatial differences in the abundance of nutrient-demanding species in upland plots in 1998 and all potential driving variables. These undoubtedly reflected similarly strong patterns of spatial correlation between driving variables in the way they have impacted British upland ecosystems in the last quarter of the 20^{th} century. This is because one of the defining features of ecological surveillance data is that responses will reflect impacts whose variation in type, intensity and prevalence are beyond the control of the observer. Hence, spatial confounding is always a strong possibility either by bad luck or because drivers are mechanistically correlated such as NHx deposition and numbers of grazing sheep. In such cases, unique signals may not be analytically attributable or point the way for further attribution studies such as those reported in section 2 of this report.

Despite these inherent design problems two important results have emerged from these analyses. Firstly, when other covariates were taken into account, the strongest correlate of presence of an atypical species in upland vegetation was average annual temperature followed by total nitrogen deposition (Table 3). However when richness of atypical species was analyzed, rather than just presence of any one member of the group, there was a clear positive link with agricultural disturbance, of which sheep grazing activity has clearly been the most important recent driver in the British uplands.

These results are interesting in that different impacts are associated with different aspects of the incursion of the species group into upland vegetation types. The presence of any species in the group appears best explained by climate and N deposition. Hence, warmer, lower altitude survey squares and those with higher N deposition are more likely to be invaded. However, the richness of the group per plot is better explained by agricultural activity. This is understandable since, grazing and improvement typify the kinds of conditions favoured by this more characteristically lowland, grassland species group.

The implications of these patterns can be summarized as a further hypothesis about the mechanism of future changes. N deposition alone seems unlikely to be able to drive marked replacement of upland species assemblages, whereas high sheep grazing pressure seems a more potent force in this respect partly because it is likely to be accompanied by other management changes such as loading of P and N fertilizer. Given the possibility of reduced stocking levels in much of upland Britain, the question may increasingly turn to past high grazing pressure and persistent effects of high N deposition as factors that may constraint recovery and drive vegetation change along other trajectories. The increasing importance of climate change clearly emphasizes the need for ongoing attribution to multiple drivers and analysis of their interacting behaviour.

3.2 Deriving botanical indicators of N deposition across upland, infertile vegetation types in Britain

3.2.1 Methods

146 higher plant species were frequent enough for testing in upland vegetation types in repeat plots recorded in the Countryside Surveys of 1978, 1990 and 1998. Analyses were constrained to plots located in heath, bog and moorland grass/mosaics in 1978 (Bunce et al. 1999). Relationships between botanical change and potential drivers were analysed using generalized linear models (proc genmod, SAS Insitute) and generalized linear mixed models (proc glimmix, SAS Insitute). The plot data were structured as sets of species presences within fixed vegetation plots nested in 1km survey squares and recorded on three occasions. For each 1km square, one value of each explanatory variable was available. Thus the intensity of the driver did not change at each successive survey. For drivers such as change in growing season length (1961-1999) and change in sheep grazing intensity (1969-2000), the explanatory variable was a rate of change, however this still applied to the 1km square and did not change with survey year. The CS square id was treated as a random, class variable. All other covariates were continuous including year of survey.

Botanical indicators of the unique effect of N deposition were sought by quantifying the strength of the correlations between spatial and temporal changes in species frequency in CS plots versus a large-scale gradient of N deposition, before and after fitting relationships with other covariates. Hypothesis tests were as follows:

Hypothesis tests

H1: Which species changed in plot occupancy over time?
Test the main effect of survey year for each species.

H2: Which drivers were significantly positively or negatively correlated with species abundance in squares irrespective of survey year.
Test the main effects of each driver against average abundance across all years of survey.

H3: Did changes in species abundance over time depend on differences in the intensity of different drivers between CS squares.
Test the interaction terms between year of survey and each driver after fitting all main effects.

H4: Which species showed the strongest correlations with N deposition when the effects of other drivers were factored out?
Test the main effect of NHx deposition against average abundance across all years of survey having first fitted all other main effects as covariates.

Covariate selection and design problems: Here we examine design and collinearity issues as they impact signal attribution in 1978 data for upland vegetation types only. These issues refer to the lack of control over the ways drivers are arranged across the sampling domain. High levels of correlation can mean that some signals cannot be attributed uniquely to drivers or that unique signals are small and omit fractions of observed variation that are shared between drivers because they have operated in spatially correlated ways. As highlighted in section 1, this can reflect drivers that are mechanistically correlated such as NHx deposition and numbers of grazing sheep, or independent yet accidentally confounded in space. Taking account of such patterns before analysis can highlight drivers that could be excluded or, if included, then expected to present foreseeable and hence understandable problems in isolating unique signals.

A number of predictably strong spatial correlations existed between drivers (Table 6, Figure 1). Sheep density in 2000, change in sheep density from 1969 to 2000 and cover of intensive agriculture were all strongly intercorrelated. Therefore, only change in sheep density was retained for further analysis, because this represents the best explanatory variable for pervasive agricultural land-use impacts in upland Britain (Fuller & Gough 1999). Change in growing season length was correlated with mean minimum January temperature and mean maximum July temperature reflecting the fact that the uplands have seen the largest recent increases in growing season length. Change in growing season length change was retained as the sole climate variable because it was the most direct correlate of climate change as an ecological driver. Modeled NHx deposition was retained as the sole explanatory variable for atmospheric nitrogen deposition because of its emergence as a more important correlate of botanical change in British infertile plant communities than modeled NOy deposition (Smart et al 2004).

Table 5. Descriptive statistics for drivers of upland species change. Based on repeat squares recorded in 1978, '90 and '98 in upland environmental zones 3, 5 and 6. See table 2 for description of drivers.

Variable	N	Mean	Std Dev	Sum	Minimum	Maximum
S2000	89	290.3	401.75	25836	0	2390
GSL	89	1.0	0.39	87	0.1	1.8
S_cge	89	10.6	14.61	941	-8.7	82.0
NHx	89	10.6	6.65	945	2.5	24.6
Tot_N	89	15.6	8.09	1390	4.9	32.5
BHint	89	15.8	27.35	1407	0	89.0
mja	89	-4.1	1.96	-362	-7.5	0.4
mju	89	19.2	2.03	1707	15.2	24.3

Table 6. Correlation matrix between drivers of upland species change.

	S2000	GSL	S_cge	NHx	Tot_N	BHint	mja	Mju
S2000	1.000	-0.221 0.0372	0.966 <.0001	0.574 <.0001	0.536 <.0001	0.437 <.0001	-0.200 0.0601	0.340 0.0011
GSL		1.000	-0.231 0.0294	-0.410 <.0001	-0.379 0.0003	-0.436 <.0001	0.328 0.0017	-0.594 <.0001
S_cge			1.000	0.614 <.0001	0.587 <.0001	0.415 <.0001	-0.279 0.0081	0.418 <.0001
NHx				1.000	0.981 <.0001	0.573 <.0001	-0.489 <.0001	0.737 <.0001
Tot_N					1.000	0.466 <.0001	-0.548 <.0001	0.690 <.0001
BHint						1.00000	-0.074 0.4927	0.576 <.0001
mja							1.000	-0.484 <.0001
mju								1.000

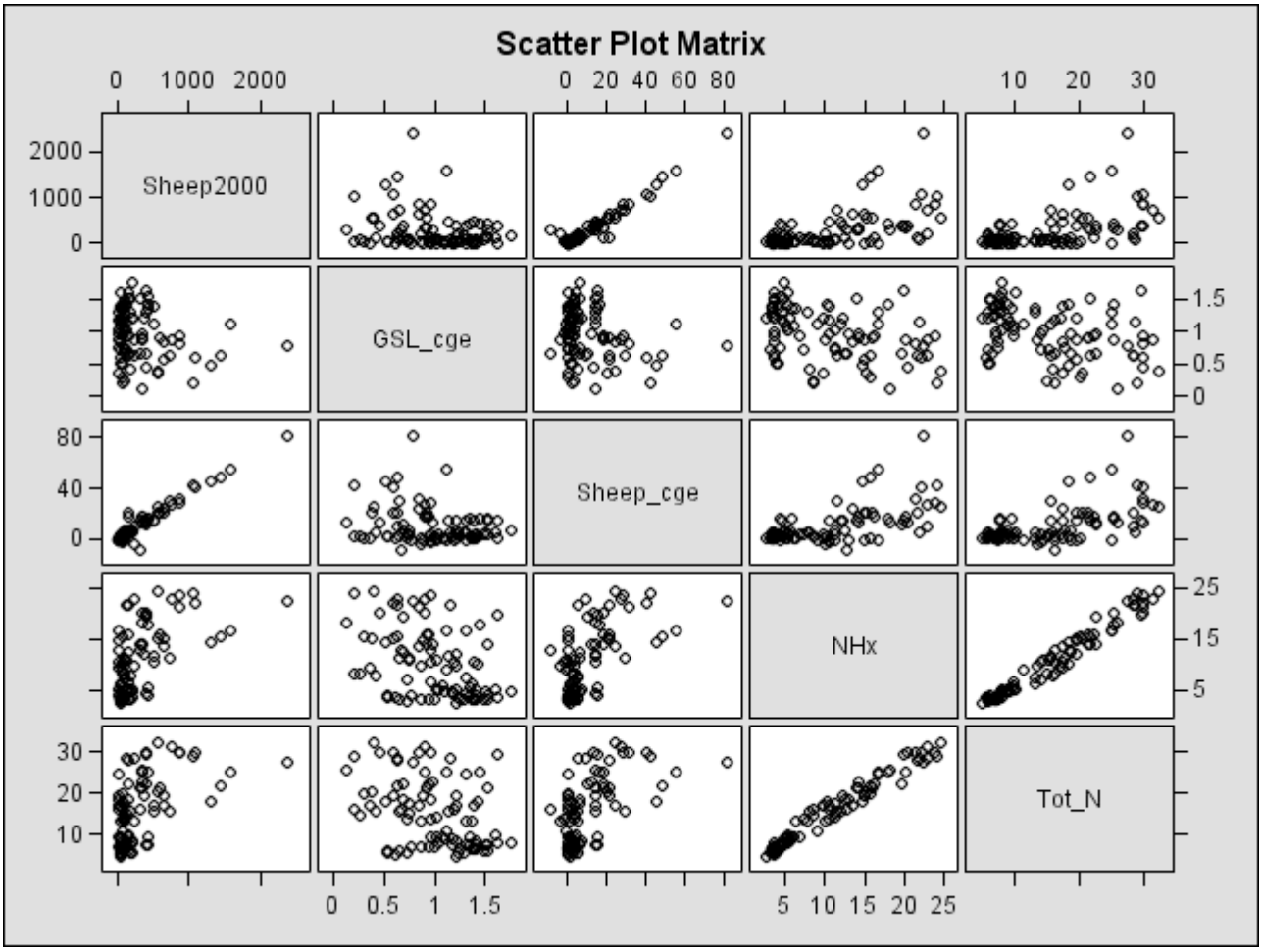

Figure 1. Matrix plot showing patterns of correlation between potential drivers of upland botanical change. See table 2 for descriptions of driving variables.

3.2.2 Results

Out of 146 species tested in upland vegetation, 61 showed significant correlations between abundance in plots averaged over all three survey years, and modeled NHx deposition in 1996. 23 of these were more likely to be present at higher deposition (positive indicators) and 38 were negative indicators (Appendix 1). 19 of the negative indicators and 6 of the positive indicators can be considered to carry the strongest, unique signal of NHx deposition impacts since they showed significant relationships with NHx even after covarying out the unique or shared signal attributable to grazing and climate change (Appendix 1 in 2006 annual report). Thus the best positive indicators include *Rumex acetosa, Deschampsia flexuosa* and *Juncus effusus*. The best negative indicators include *Calluna vulgaris, Erica tetralix, Carex echinata, Pinguicula vulgaris* and *Erica cinerea*. The plausibility of the indicators is supported to some extent by the fact that positive indicators generally have higher Ellenberg N values and the negative group lower Ellenberg N values (Figure 2). The identification of these indicators was based on many multiple tests and an important caveat is that some significant results may have occurred by chance purely because many tests were carried out.

Links between NHx deposition and changes in abundance between surveys were explored by first testing for overall change and then determining whether any of the observed change between 1978 to '90 to '98 was explainable by NHx deposition ie. did species increase more or decrease more where NHx deposition was higher? Overall significant changes up or down are indicated in column 6 in Appendix 1 to the 2006 TU annual report. If change was additionally explained by NHx deposition then this is indicated by the presence of an estimate of the size of the effect in the Type I Yr*NHx estimate column. The results indicate that very few significant temporal changes in indicator abundance were explained by NHx deposition. This is not surprising since the power of such tests is low. It also implies that much of the botanical change correlated with spatial differences in NHx deposition had happened before 1978 as well as during the 20 year survey interval.

By analyzing relationships between individual species abundance and each potential driver having covaried out the shared effects of other drivers, we can tell which driver appeared to generate the strongest unique signals in upland GB based on the number of species that had significant partial correlations with each driver. Our tentative conclusion is that NHx deposition generated more species specific signals than either change in growing season length or change in sheep grazing intensity (Figure 3). This could simply reflect the fact that the gradient of NHx deposition was longer and better replicated than the others across Britain. However, descriptive statistics and correlations between drivers (Tables 7, 8 and Figure 1) suggest that the other gradients were not appreciably shorter or less variable.

3.2.3 Discussion and implications

Despite the fact that the scale at which driving variables were resolved was much bigger than CS sampling plots, modeled NHx deposition displayed significant correlations with 41% of the common higher plant species present in upland CS plots in the three survey years. A smaller subset (17%) carried the strongest unique signal of NHx deposition. Overall, more species-level differences in abundance between CS squares could be attributed to spatial differences in NHx deposition than to change in sheep density or growing season length.

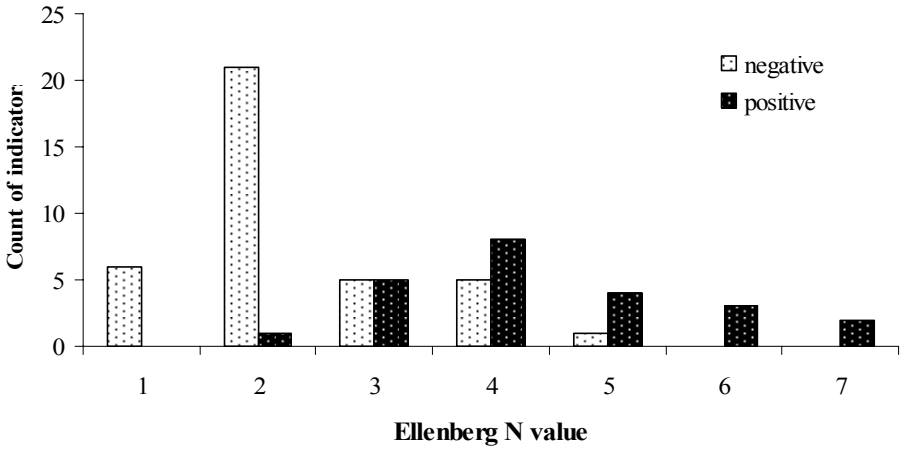

Figure 2. Count of plant species whose abundance in CS plots in 1978,'90 and '98 showed either a negative or positive correlation with modeled NHx deposition in 1996. Indicator species have been grouped by their Ellenberg N value.

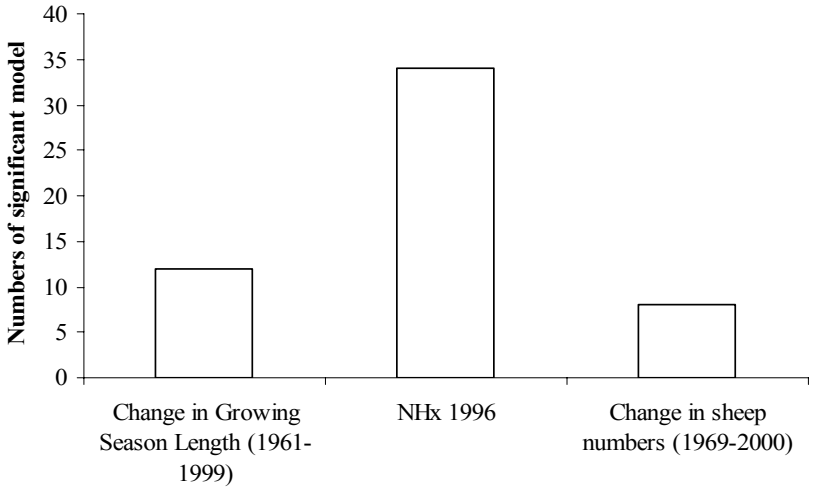

Figure 3. Numbers of species out of 146 tested, whose average abundance in CS plots in 1978, '90 and '98 showed significant partial correlations with three key drivers of upland botanical change.

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Work Package 2:
Impacts, Recovery and Processes

Task 16
Model-based risk assessment of the vulnerability of rare coastal species to N deposition

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Task 16 – Model-based risk assessment of the vulnerability of rare coastal species to N deposition

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1. Summary

Four rare sand dune species (*Equisetum variegatum, Petalophyllum ralfsii, Centaurium littorale* and *Parnassia palustris*) were selected for modelling the likely effects of nitrogen deposition on rare species. Twelve UK sites were surveyed for community composition and soil data associated with the target species; control locations at each site were surveyed where the target species did not occur. Species-environment models were built up for each species using binary logistic regression. These were then used to predict changes in the probability of occurrence under a scenario of a + 10 kg N ha^{-1} yr^{-1} increase in N deposition over 50 years. Changes in the soil N parameters were estimated using a simplified N accumulation model under this scenario and in combination with the species-environment models, a ranking of the sensitivity of the four species to changes in nitrogen deposition was produced. In order with most sensitive first the species were: *Equisetum variegatum* (strong decrease in probability of occurrence), *Centaurium littorale* (no change), *Parnassia palustris* (slight increase), and *Petalophyllum ralfsii* (increase).

2. Policy Relevance

Sand dunes are a Priority BAP habitat, and contain over 70 red data book, nationally rare or scarce species, listed in Annex 2 of the Habitats Directive, as well as other species of conservation importance. Many of the rarer species occur in early successional habitats and are highly susceptible to N-deposition driven soil and vegetation change. Until now, very little has been known about the likely effects of N deposition on rare species in any UK habitat. While many factors contribute to rarity, this study has provided the first data predicting how sensitive some rare species are to changes in the soil bio-chemistry caused by increased N deposition, using dune slacks as a model habitat.

3. Objectives

- Evaluation of the potential chemical and biological impacts of nitrogen deposition in coastal systems

4. Methods and results

4.1 Methods

Four rare species were selected out of a final list of six potential species identified in the preparatory phase of the study. Selection criteria were that species were UK Red Data Book, Nationally rare or scarce, preferably sand dune obligates, but that had a reasonable

geographical distribution across the UK. Species with too restricted a distribution were not selected, in order to better determine their soil requirements across a number of locations, and to determine the potential impacts of N deposition. The four species chosen were the liverwort *Petalophyllum ralfsii*, the horsetail *Equisetum variegatum*, and the vascular plants *Centaurium littorale* and *Parnassia palustris*. These are all species of dune slacks or damp grassland, although *Parnassia* also occurs in other habitats than sand dunes and is slightly more widespread than the other species. Twelve sites around the UK were chosen where some or all of these species occurred. The fieldwork campaign was conducted in late summer/early autumn 2006 in order to maximise the chances of finding *Petalophyllum* but still being able to find and identify the vascular plants. Occurrences of the four species at the twelve sites, plus the N deposition to each site derived from the APIS system, are shown in Table 1.

Sites were surveyed from North to South to account for seasonal effects at the end of the year. At each site, approximately 6 locations per target species were surveyed, recording species abundance in 2 x 2 m quadrats, together with 6 locations in similar habitat but where the target species did not occur which were surveyed as controls. Other variables recorded in the field were slope, percentage bare ground, sward height, and presence and type of grazers. Within each quadrat soil samples were taken (5 cm diameter cores, to 15 cm depth). The soil samples were processed back in the laboratory using a novel technique for assessing mineralisable N. The cores were brought to field capacity by flushing with a known volume of de-ionised water, incubating at 10 °C for 3 weeks then extracting for available N. In addition, conductivity of the flushed leachate solution was measured together with the following determinands post incubation: moisture content, loss on ignition, soil conductivity, pH (in water and in $CaCl_2$), total N, total C, and C:N ratio.

Table 1. Rare species occurrence, survey sites and N deposition

Site name	Approx. Grid Ref	N deposition (kg N ha^{-1} yr^{-1})	*Petalophyllum ralfsii* Red Data Book. Schedule 8 sp.	*Equisetum variegatum* Nationally Scarce	*Centaurium littorale* Nationally Scarce	*Parnassia palustris* Local distribution
Achnahaird	NC016135	4.2	√			
Morrich More	NH851836	6.3		√		√
Tentsmuir	NO490280	10.2			√	√
Aberlady	NT463821	11.1		√		√
Holy Island	NU112433	10.2	√	√	√	√
Ainsdale	SD300116	19.5	√	√	√	√
Talacre	SJ111850	11.1	√			
Newborough	SH427629	11.1	√	√	√	√
Ynyslas	SN608941	8.7	√	√		
Whiteford Burrows	SS447959	11.3	√	√		
Scolt Head	TF708447	18.8	√			
Dawlish Warren	SX986789	14.1	√			

4.2 Data analysis

Mean Ellenberg scores for N (nutrient), L (light), R (pH) and F (moisture) were calculated for each quadrat. The quadrats were coded as to presence/absence for each target species and analysed using binary logistic regression in Minitab v.14.1 with a Logit function. All primary variables (i.e. excluding Ellenberg scores) were included in the initial models including

quadratic and cubic terms, and the best model for each species was fit by backwards elimination of variables, usually eliminating terms not significant at p = 0.10 although some terms with a poorer fit were retained, primarily those associated with soil N content or N mineralisation. In the final equations, the models converged successfully producing equations which significantly explained the probability of occurrence for all four species.

4.3 Results

The fit to the equations are shown in Table 2 below. The best model fit occurred with *Petalophyllum* as shown by the high Somers' D statistic. The variables explaining their variation are very different between species. With the two vascular species there was little difference between fit of the primary variable and the Ellenberg models. The primary variable model fit much better for *Petalophyllum* while the Ellenberg based model fit much better for *Equisetum*. While all models were significant, the primary and the Ellenberg models did not always select comparable variables. For example, the *Petalophyllum* Ellenberg model was only significant for pH (Ellenberg R) which was not significant in the primary variables model. Nonetheless, it appears that Ellenberg values successfully produce alternative predictive models in place of measured soil variables for these species.

Table 2. Variables in equations and fit of models, including alternative models based on Ellenberg values for the four rare species

Species	Equation	G	p	Somers' D statistic
Petalophyllum ralfsii	e^(-0.163172 VegHeight + 0.0014681VegHeight^2 - 0.0000038VegHeight^3 + 0.298002 %Sand - 0.011438 %Sand^2 + 0.0001131%Sand^3 - 96.7012 %N + 784.31 %N^2 - 1731.04 %N^3 + 4.62889)	72.631	< 0.001	0.80
P. ralfsii alt.	e^(4.04719 Eberg_R - 25.6461)	25.955	< 0.001	0.49
Equisetum variegatum	e^(-179.952 pH_Water + 24.7854 pH_Water^2 - 1.12719 pH_Water^3 + 0.0020157Leachate_conductivity + 0.374184 Coarse_root_biomass - 1.29222 CNratio + 0.109586 CNratio^2 - 0.00263 CNratio^3 + 432.998)	26.079	0.001	0.46
E. variegatum alt.	e^(-4.36033 Eberg_N + 1.4067 Eberg_R + 4.99337 Eberg_F - 5.24251 Eberg_L + 16.2169)	73.264	< 0.001	0.76
Centaurium littorale	e^(0.0225685MineralisableNH$_4$pergramOM + 29.738 %N - 128.829 %N^2 + 48.1516 %N^3 - 3.67828)	10.698	0.030	0.41
C. littorale alt.	e^(-1.1712 Eberg_F + 4.11369)	13.452	< 0.001	0.47
Parnassia palustris	e^(0.0220575VegHeight - 0.0000487 VegHeight^2 + 253.401 pH_Water - 34.5224 pH_Water^2 + 1.56363 pH_Water^3 + 2.41879 Coarse_root_biomass - 1.01514 Coarse_root_biomass^2 + 0.109506 Coarse_root_biomass^3 - 0.0016017 MineralisableNH$_4$pergramOM^2 + 0.0000235 MineralisableNH$_4$pergramOM^3 - 622.06)	28.326	0.002	0.55
P. palustris alt.	e^(-0.830921 Eberg_N + 1.16704 Eberg_F – 5.12498)	26.488	< 0.001	0.61

With the logistic equations for each species established, their sensitivity to changing N deposition could be assessed. The equations based on primary variables were used for further modelling steps. In order to best predict the effects of N deposition on N accumulation in soil and changes in the C:N ratio for sand dunes it would be necessary to fully parameterise the MAGIC model for dune systems. This was not part of the current contract but may be part of the next phase of the Terrestrial Umbrella. In the meantime a simplified version incorporating the basic components of the MAGIC model was used to estimate sensitivity of these species.

It has been argued that, for tropical soils at least, physical soil structure (particle size, sand/clay fractions etc.) determines the maximum possible carbon accumulation for any soil type (Feller & Beare, 1997). If it is assumed that sand dune soils are in equilibrium and that they have reached carbon saturation then the effects of changing N deposition can be calculated based on a calculation of the annual N retention of atmospheric N inputs with respect to current N and C stocks in the soil. This also assumes that the proportion of N leached (and other losses such as denitrification) remains constant as N content increases. Using this method, changes in occurrence probability for each of the four species were assessed using a simple scenario of an increase in N deposition of + 10 kg N ha^{-1} yr^{-1} on top of current deposition over a 50-year time period. Therefore, at the sites where each target species occurred mean values were calculated for N deposition, %N content, N pool and C pool in order to predict change in probability of occurrence. Leaching losses from dune slacks were taken from Jones *et al*. (2005) as an estimate of the proportional retention of the additional atmospheric N.

One or more of the soil N parameters was included in all the models, therefore the sensitivity to changes in N deposition could be assessed for all four species. In the case of *Petalophyllum*, whose model was partly driven by changes in the %N content of the soil, the prediction was for a 26 % increase in the probability of occurrence of this species. The model for *Equisetum* was driven partly by changes in C:N ratio and predicted a 55 % decrease in the probability of occurrence. The model for *Centaurium* was driven both by changes in %N content and mineralisable NH_4 per gram OM. While we can theoretically calculate changes in %N content, our process understanding of available N or mineralisable N dynamics in dune systems is very poor. Therefore, in order to estimate impacts of increased atmospheric N inputs, the relationship between mineralisable NH_4 per gram OM and N deposition was established over the deposition gradient surveyed in this study. When soil pH and moisture content (using Ellenberg F as a proxy) were used as covariates in the regression, there was a weak negative trend of decreased mineralisable NH_4 per gram OM with increasing N deposition ($p = 0.061$). The negative trend was confirmed using mineralisable NH_4 as the sole regression term. Therefore this relationship was used to predict levels of mineralisable NH_4 per gram OM under the future N deposition scenario (slightly outside the range of values observed here:- 22.75 kg N ha^{-1} yr^{-1} compared with a maximum N deposition of 19.5 kg N ha^{-1} yr^{-1} in the survey). From these calculations, there was minimal impact on *Centaurium* of the modelled increase in N deposition, with a slight increase of 4 % in the probability of occurrence. The same relationship between mineralisable NH_4 and N deposition was used for the prediction for *Parnassia*. This suggested an 11 % increase in the probability of occurrence of this species.

Therefore, the species ranked in order of decreasing sensitivity to negative effects of N deposition are: *Equisetum variegatum, Centaurium littorale, Parnassia palustris* and *Petalophyllum ralfsii* (Figure 1).

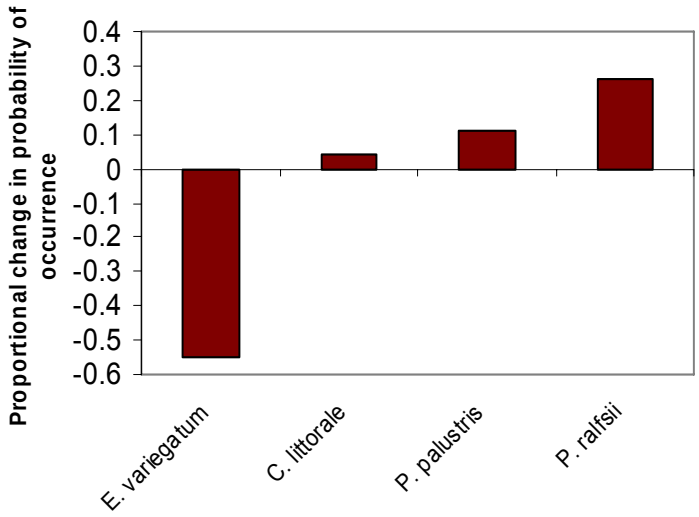

Figure 1. Modelled proportional change in probability of occurrence under scenario of increased N deposition (+10 kg N ha^{-1} yr^{-1} above current background).

Some further analysis aimed to ascertain wider impacts of N deposition on the species and soil characteristics of dune slacks along this deposition gradient. Relationships with N deposition were tested using linear regression, using soil pH and moisture content in the regression in order to correct for potentially confounding factors. Mean Ellenberg F values were used as a proxy for soil moisture content as the soil cores had been brought to field capacity as part of the mineralisation study. There were no significant correlations of species richness or any of the soil parameters with N deposition, although there was a non-significant trend towards decreased mineralisable NH_4 per gram OM with increasing N deposition.

5. Discussion and implications

This work has provided valuable predictions on the effects of N deposition on rare species in dune systems, and has moved a step closer to establishing the vegetation and soil characteristics required for persistence of these species. Of necessity, the modelling has had to make various assumptions which may not be valid for dune soils. The principle assumptions centre on the effect of continuing N deposition on soil N and C contents and the associated soil N parameters, which have been assumed to increase linearly while carbon stocks remain constant. Firstly it is highly unlikely that C stocks remain constant with increasing N deposition, since dune soils are relatively young and are unlikely to be in equilibrium. Secondly, work by Jones *et al*. (2007) has shown that N (and C) accumulation are not linear but follow a sigmoidal curve in the first 60 to 100 years of soil development. Therefore, the rate of N accumulation partly depends on the point on curve, which is particularly important for the very early successional soils where *Petalophyllum* is often found. The rate of N accumulation slows considerably once the plateau of the sigmoidal curve is reached. Thirdly, the proportion of inputs lost to leaching is likely to increase as %N content increases rather than remain constant. Therefore the assumptions made in this study will tend to under-estimate the likely effects of N deposition for plants growing on the

youngest soils, but will over-estimate effects for species on older soils. Despite these limitations, the study provides an estimate of the relative sensitivity of these species under a worst-case scenario, and so is valid as a first approximation of the possible effects of N. It should be stressed that N deposition is only one factor affecting species abundance and other factors such as climate and site management (e.g. maintaining low sward height through grazing or mowing) have a greater impact in the short term.

These results suggest that only one species, *Equisteum variegatum,* will experience a direct negative impact as a result of increased N deposition, while the other three species show either little change or their probability of occurrence may increase. These predictions are based directly on the nitrogen chemistry of the soil however, and secondary effects of N deposition on the surrounding vegetation such as increased biomass and sward height (Jones *et al*. 2004) may lead to increased competition from other species. What this study shows therefore is that the latter three species are not directly negatively affected by N deposition, but further work is necessary to assess indirect impacts of nitrogen-mediated competition on these species.

Preliminary outputs from this work were presented at the CAPER conference in 2006. It is aimed to produce at least one paper, to be submitted to 'Ecological modelling'. Data from the field survey has been stored centrally with CEH.

6. Possible future work

The next important step for this work is to parameterise the MAGIC model for dune systems. Many of the soil physical measurements have already been taken but there remains a need for accurate leaching data under a range of N deposition levels. It is only by measuring how much of the extra N is retained in the system and estimating the effects on biological N fixation that the true rate of N accumulation can be modelled. There is also a need to further develop the community composition (GB MOVE) modeling. This will enable the more subtle indirect effects of competition resulting from increased N deposition to be assessed.

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Work Package 3:
Dynamic Modelling and Vegetation Response

Task 17:
Vegetation module for dynamic modeling

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Task 17 - Vegetation module for dynamic modeling

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1. Summary

- Updated and finalized empirical niche models (GBMOVE) have been produced for 318 bryophytes and 851 higher plants species. Initial testing on an independent dataset of paired soil and plants species presence data showed good performance for an arbitrary subset of 9% of species models. Performance was very good when models were solved using observed mean Ellenberg values but less good when solved using observed soil data. This was due to the uncertainty introduced by the calibration functions that translate measurements of soil pH, %C, %N and soil moisture into mean Ellenberg N, R and F indices.

- A new set of calibration equations were produced using a multiple regression approach. These achieved a significant improvement in r-sqrd value when explaining mean Ellenberg indices. While these improved functions still reduce the accuracy of niche models, when used to translate soil conditions into species composition they are a vital link between dynamic soil model outputs and GBMOVE niche models.

- Predictions of soil pH and C:N ratio for Countryside Survey field plots in 1978 and 1998 were generated using the VSD dynamic soil model. Comparisons with observed soil data showed a lack of fit and emphasize the pressing need for an analysis of the sources of uncertainty involved. It is vital, for example, to understand how much uncertainty is being contributed by modeled N and S deposition errors, unrepresentative soil chemistry data, and the influence of unmeasured drivers of fine-scale soil and vegetation change.

- We suspect that the use of the average values for each soil and vegetation type combination in the model served to flatten the range of model outputs. This is more acceptable if the goal is to model average conditions at a coarse, landscape-scale. Therefore a key conclusion is that VSD (or MAGIC) should only be used as regional-scale assessment tools unless better soil data is available to parameterise the model more specifically to a particular area

- In readiness for GB-level regional predictions of change in plant species habitat suitability, a method has been devised and tested for selecting species pools for each 1 km square. Each species list then reflects those indicators present on a modeled site or within the particular 1km pool that can be modeled using the linked soil and vegetation modules.

- In recognition of the importance of testing scenarios of biodiversity change driven by multiple drivers, we have continued to develop a second generation of niche models that include climate variables and their interactions with abiotic gradients. These require further development and testing.

2. Objectives

Agreed Milestones (M) and Deliverables (D)

M1.1	*Incorporation of climate variables in new regression surfaces*	*Oct 2004*	**Complete**
M1.2	*Completion of species/area curve correction factors for different vegetation sampling scales*	*Mar 2005*	**No longer applicable**
D1.1	*Report new regression surfaces and species/area correction.*	*Mar 2005*	**No longer applicable**
M1.3	*Development of spatially explicit species-pool definitions*	*Oct 2006*	**Complete**
D1.2	*Report species-pool selection procedures*	*Mar 2006*	**Complete**
M1.4	*Site-specific model testing*	*Dec 2006*	**Complete**
M1.5	*Regionalised model testing*	*Mar 2007*	**Complete**
D1.3	*Final model projections & reporting*	*June 2007*	**Incomplete**

The aim of this work package is to develop a general capability for modeling atmospheric pollutant deposition impacts on terrestrial plant species. The approach has been to produce empirical niche models describing the abiotic and climatic preferences of as many higher and lower British plant species as possible. These models, called GBMOVE, are then solved using values of abiotic and climatic variables that themselves change to reflect a scenario of environmental change (Fig 1). Since the main policy driver has been analysis and modeling of the impact of changing S and N deposition, the emphasis has been on driving change in habitat suitability for plant species by in turn modeling change in soil conditions using the dynamic models VSD and MAGIC. Both models output time series of soil variables (pH and C:N ratio) that can be used to solve the plant species niche models at each time step (typically annual to decadal) at scales that range from a vegetation patch within a specified nature reserve through to averaged scenarios for spatially explicit 1km squares. Greater accuracy at small scales appears to depend critically on representative soil chemistry data and probably accurate N and S deposition data. Projections of change in habitat suitability can then be achieved and are most meaningful if modeled species are either known to be present in a 1km square or within a specified site, and are acknowledged indicators of good or bad ecological condition, for example Common Standards Monitoring indicators[5].

During this work program niche models have been completed for a large number of plant species. Additional work has also advanced on a number of fronts all geared toward developing a general yet accurate capability for modeling pollutant impacts by combining species and soil models. Key activities have included i) improving the way the niche models are linked to soil model output by developing better calibrations between soil measurements and vegetation (mean Ellenberg) indices of abiotic conditions, ii) testing the separate components of the growing model chain (soil model predictions, calibration equations and niche model predictions) at site and national scales , iii) developing a way of selecting a local species pool to be modeled that reflects the species composition in any one 1km square across Britain, iv) finalizing the plant species niche models by building in climate variables and deriving an updated and complete set of species *pmax* values. These are the maximum probabilities of occurrence for each species. They are found where environmental conditions

[5] www.jncc.gov.uk/pdf/CSM_06species.pdf

are optimal for the species concerned and are required in order to factor out differences in species abundance at the environmental optimum.

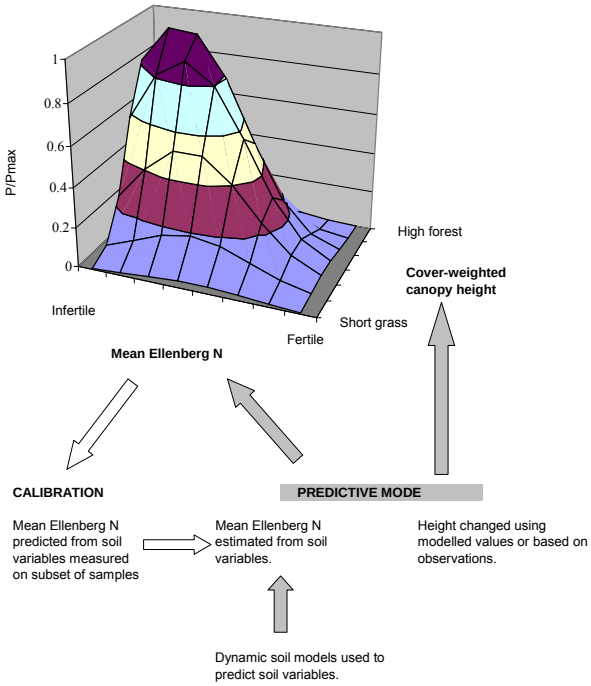


Figure 1. Steps in the construction and application of niche regression models for British higher and lower plants. Two abiotic axes (soil fertility and disturbance rate) are shown from the response surface for the shade-tolerant woodland herb *Oxalis acetosella*.

The surface was evaluated at soil pH = 4.7 and 45% soil moisture. The logistic regression probability (y-axis) is interpreted as an index of the favourability of abiotic conditions for the species. This is the result of solving a multiple logistic regression with mean Ellenberg N, F and R values plus cover-weighted canopy height as explanatory variables. To standardise across species, predicted probabilities are rescaled by dividing each by the probability (*pmax*) at the abiotic optimum of each species. In predictive mode each regression model is solved given new input values of explanatory variables. Translation of new input values of soil %C, %N, soil pH and soil moisture into mean Ellenberg values is achieved using calibration equations originally generated from the subset of quadrat data with paired species-composition and soil data (see text). New input values of soil variables can originate from measurements or from the output of a biogeochemical model run over a series of time-steps.

3. Further development of GBMOVE models for British plants

3.1 Model building

3.1.1 Methods

Multiple logistic regression was used to construct equations that could predict habitat suitability for as many higher and lower plants as possible, based on their abundance along key environmental gradients defined by mean Ellenberg values. Each model used presence/absence data for each plant species in each plot as the response variable to be modelled. Ellenberg values for higher plants were based on the list recently updated for the British flora (Hill et al 1999). An average Ellenberg index could therefore be calculated for any quadrat based on the number of species and hence individual values present. In addition, mean cover-weighted canopy height was used as an index of the disturbance/successional status of the vegetation in each quadrat. This provides a niche gradient that tracks species' responsiveness to management intensity in the form of biomass removal . Average canopy height for each species was based on values taken from Grime et al (1995) and Stace (1997). Bryophytes were excluded from these calculations.

To avoid circularity during model construction, Ellenberg values of the species being modeled were excluded from the calculation of the mean Ellenberg index. Only if this is done is the index (explanatory variable) independent of the species presence (response variable).

Variable selection was carried out by first testing the explanatory power of each variable separately (both linear and quadratic terms) and then entering all those that were significant, and their interactions, into a stepwise elimination procedure (SAS proc logistic). Significance was tested using likelihood ratio tests. Where quadratic or interaction terms were significant then the contributing linear terms were also included by default. All analysis and data manipulation steps were carried out using SAS procedures and the SAS macro language. Model assessment was carried out by visual inspection of the fit of each predicted response curve to the histogram of percentage plots occupied by each species in intervals along each abiotic gradient. To allow examination of each model response curve, values of explanatory variables for gradients not being inspected were set to the mean value among occupied plots in each interval of the inspected gradient.

Across British ecosystems, a number of species are known to exhibit bimodal response curves along particular gradients but especially substrate pH (Roy et al 2000; Hill et al 1999). For these species, third and fourth order terms and their interactions were also tested and retained where significant.

3.1.2 Results

Of 2059 higher and lower plant species recorded in the training dataset, 1253 occurred in more than 10 quadrats. Of these a multiple logistic regression model with at least one significant explanatory variable could be constructed for all except 11 species. A further 75 species were modelled but then rejected from further validation and testing. These species have a coastal distribution (defined following Hill et al 2004). Because we lacked quadrat-scale data on salt load either from the atmosphere or through inundation, these models currently lack a quantitative expression of response along this important gradient and were

considered of limited use. In total, models were produced for 318 bryophytes and 851 higher plants.

Species fits to the training data were inspected visually focusing on CSM indicators in the first instance. These showed that important niche attributes such as wide and narrow tolerance about the optimum, skewed optima and bimodality could all be readily captured by the relatively simple MLR model formulation (Fig 2). While these graphs also indicate a good fit to the observed training data, more rigorous testing comes from comparing predictions against observations from independent data. Such testing has been carried out for 9% of the plant species modeled (see below) but this critical activity remains a long-term campaign to be carried out as independent soil and plant species data accumulate.

Table 1. Quadrat datasets used to build GBMOVE regression models for British higher and lower plants.

Data source	**Number of plots**
Key Habitats Survey (1992)	548
Countryside Survey (1998)	7221
NCC/ITE Broadleaved woodland survey (1971)	1648
National Vegetation Classification	30049
Total	32483

Table 2. Calibration equations that predict values of mean unweighted Ellenberg values from four soil variables; MC (%soil moisture content), soil pH, C (% carbon) and N (% nitrogen). See text for further details.

Unweighted mean Ellenberg score	**% variance explained**	**Equation**
F (wetness)	69.7	=exp(1.5661+0.004046(MC))
R (substrate pH)	77.9	=0.5293-0.02503(MC)+1.665(pH)-0.1061(pH^2)-0.00566(C)
N (substrate fertility)	78.2	=exp(0.7751-0.00006(MC)-0.00009(MC^2)-0.01475(C)+ 0.000099(C^2)+0.2639(pH)-0.01684(pH^2)+0.1908(N))

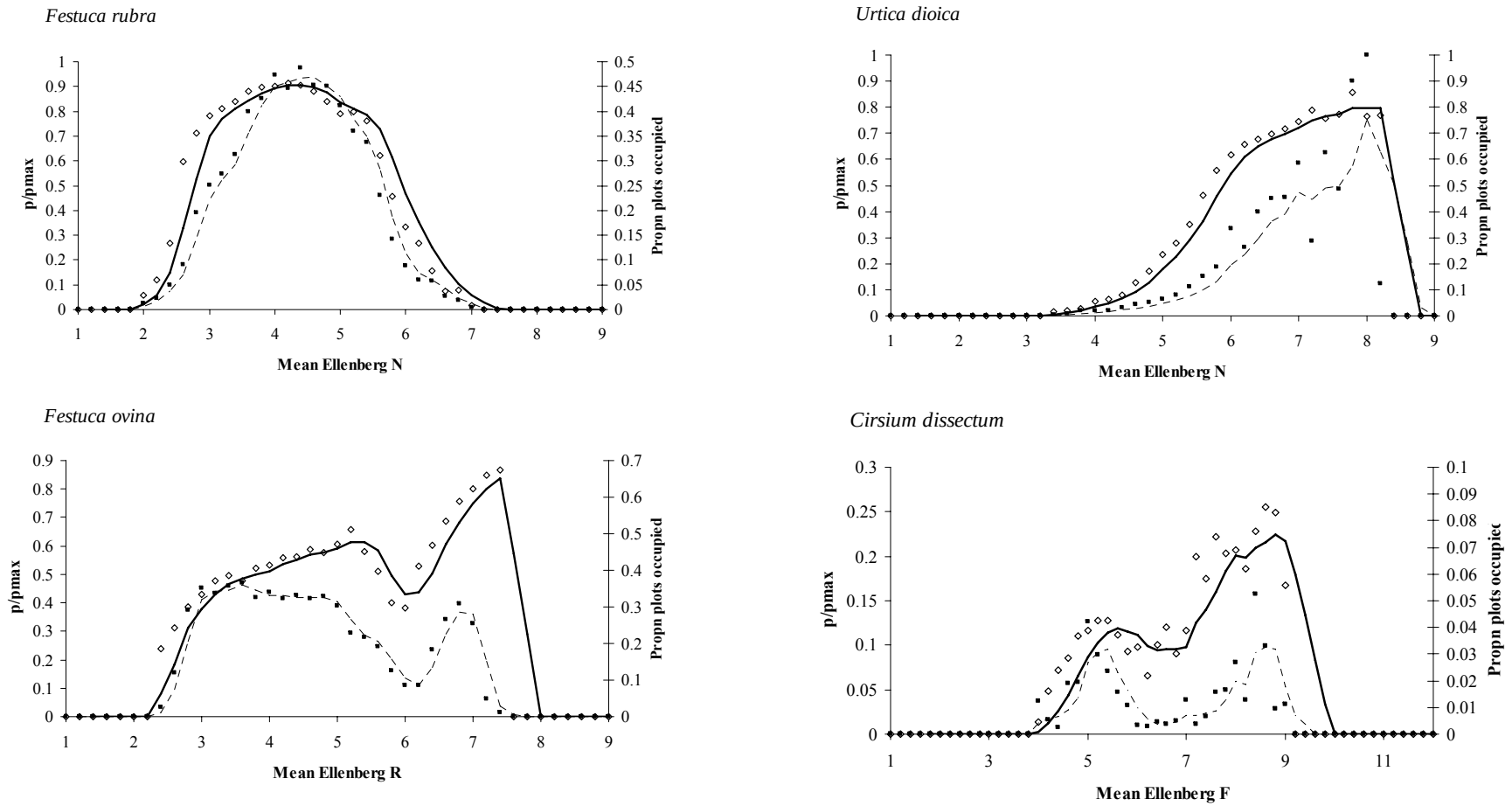


Figure 2. Species models fitted to the GB training dataset. Open diamonds and the solid line (moving average) show the model prediction of habitat suitability along each gradient. MLR models were solved at intervals of 0.2 Ellenberg index units along the x-axis. Terms for other abiotic gradients were then solved in each interval using the average values for other indices in plots occupied by the species concerned. In each graph, probabilities were rescaled by the *pmax* value for each species. Black squares and the dashed line (moving average) indicate the proportion of plots in each interval actually occupied by the species.

3.2 Calculation of model pmax *values*

3.2.1 Methods

At the scale of small vegetation sampling plots, species that are inherently less frequent even when abiotic conditions are optimal, will have lower predicted maximum probabilities than more abundant species. Hence, differences in probability of occurrence between species will reflect overall patterns of scarcity in the training dataset as well as reflecting the favourability of abiotic conditions. If predicted probabilities are divided by the maximum probability (*pmax*) possible then these effects are removed thus enabling probabilities to be used as standardised indices of habitat suitability. Two methods were used to determine *pmax* values: A numerical solution was firstly developed (see 2005 TU Annual report). This approach highlighted particular model forms where no unique optimum and hence *pmax* value, could be calculated. An analytical solution was therefore applied where, for each species model, a single *pmax* value was found by a data-intensive search of the entire occupied niche space for each species. To do this, each of the four abiotic gradients was divided into intervals and species probabilities generated for every combination of interval values across all gradients.

3.2.2 Results

Pmax values were determined for all 1169 species models. As expected *pmax* showed a significant positive correlation with frequency in the training dataset (Spearman rank correlation = 0.57, $p<0.0001$) but the form of the relationship was characterised by markedly heterogenous variance such that high or low *pmax* values were associated with low species frequency but the most frequent species were more likely to be associated with high *pmax* values.

3.3 Calibration equations

3.3.1 Methods

To produce regression models for plant species, paired soil measurements and presence/absence data is needed for all the species concerned. However, soil data is not available for all plots in which species occurred. Because Ellenberg values are available for most plants, a mean index can be created for each plot and each key gradient. To be able to link these mean Ellenberg indices to soil variables the one must be calibrated in terms of the other. Once this is done, we can use the calibration equations to predict mean Ellenberg values in terms of soil variables. Then, because mean Ellenberg values are the explanatory variables in each niche model, changes in their value can be solved to give changes in habitat suitability for each species.

Multiple regression was used to calibrate mean Ellenberg indices in terms of five soil variables - %organic carbon (%C), % organic nitrogen (%N), % soil moisture, soil pH and Olsen's extractable phosphorous. All were measured in a subset of the quadrats (n=1033) in the Countryside Survey 2000 dataset. Calibration equations were constructed by selection of those soil variables that minimised unexplained variation in the mean Ellenberg indices having taking into account the total number of model parameters generated. First, all candidate variables were tested as the single predictor for each mean unweighted Ellenberg index and non-significant terms were rejected. The best minimum adequate model (MAM)

was then determined by manually fitting sequences of the remaining soil variables plus their quadratic and interaction terms and comparing the performance of pairs of simple and more complex nested models by deviance reduction (F) tests (e.g. Manning et al 2005) This process was carried out for $\log_e$, logit and untransformed response variables. Because the calibration dataset of paired soil measurements and mean Ellenberg indices consisted of quadrats nested in the 1km sample squares of the Countryside Survey, the 1km square was treated as a random effect in a general linear mixed model (GLMM). The degrees of freedom attaching to each model were downweighted according to the approximation of Satterthwaite (1946). Analyses were carried out using restricted maximum likelihood estimation implemented in SAS (Littel et al 2000).

The importance of each explanatory variable in each final model was expressed by determining the unique (partial) contribution of each variable in the presence of all other selected variables and then comparing this to its contribution when entered as the only variable in each model (see Singer 1998 for details).

The construction of calibration equations differed from previous approaches (e.g. Ertsen et al 1998) in that multiple soil variables were allowed to predict each mean Ellenberg index rather than a calibration relationship sought from a hypothesised link between a single dependent soil variable and a single mean Ellenberg value; for example attempting to model soil C/N in terms of just mean Ellenberg N (fertility) or soil moisture in terms of just mean Ellenberg F (wetness). Here the mean Ellenberg value is treated as the dependent variable which we seek to model in terms of multiple explanatory soil variables. This is consistent with viewing Ellenberg indices as composite summaries of each gradient and is particularly appropriate for fertility where multiple factors control net primary production so that mean Ellenberg N is often best correlated with biomass yield rather than availability of individual macro-nutrients (Hill & Carey 1997, Schaffers 2000, Smart et al 2002). Another advantage of this approach is that it helps counter the problem of missing terms in the niche models. Each model is constructed from the automatic, stepwise selection of mean Ellenberg indices as explanatory variables. If two Ellenberg-defined gradients have high shared explanatory power for a species presence then one may be deleted by the stepwise procedure. Thus, if each mean Ellenberg index were only defined by one pre-determined soil variable such as %N, and that index was excluded from a species model, then changes in %N, could never influence the probability of occurrence of that species. However, if mean Ellenberg indices are calibrated against more than one soil variable so that, for example, %N and soil moisture can appear as explanatory variables for more than one Ellenberg index, then change in %N and soil moisture can still influence predicted probabilities even if the most logically related Ellenberg index was not selected as a model term.

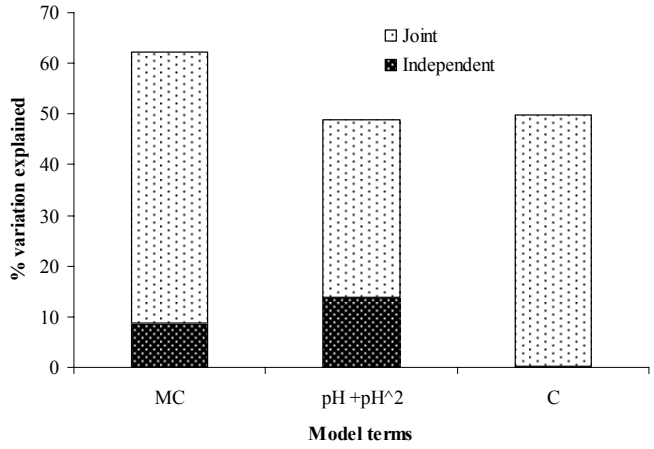
3.3.2 Results

Calibration equations were constructed between all three mean unweighted Ellenberg scores and soil variables (Table 2 and Fig 3). The percentage of observed variation explained was moderately high (Table 2). Ellenberg F was best explained solely by % soil moisture content while selection of the best minimum adequate model vindicated treatment of mean Ellenberg R and N values as composite indices best explained by multiple soil measurements (Table 2). Mean Ellenberg R was best explained by a combination of % soil moisture, soil pH and % carbon. Examination of the independent versus partial contribution of selected variables showed very considerable overlap so that the total explained variation mainly comprised joint variation in %C, %soil moisture and pH (Fig 3a). This reflects their spatial intercorrelation

across the training dataset, in part related to obvious mechanistic relationships between for example, soil wetness promoting organic matter accumulation in turn often associated with low base status and low pH. The highest unique explanatory power was attributable to substrate pH (linear + quadratic term) as expected if Ellenberg R values were designed to track a substrate pH gradient.

Mean Ellenberg N was best explained by % soil moisture, soil pH, % carbon and % nitrogen (Fig 3b). Again, the majority of the explained variation was shared between predictors, whilst soil moisture content (linear+quadratic terms) had the highest unique explanatory power (Fig 3b). Overall therefore, the explanatory power of any one variable could not be separated from the others. As for mean Ellenberg R, this undoubtedly reflects mechanistic causal links between soil moisture, carbon and nitrogen content and pH. Despite Ellenberg N values originating as an ostensible indicator of N availability, %N had the lowest unique explanatory power of all terms selected for the final model.

a.



b.

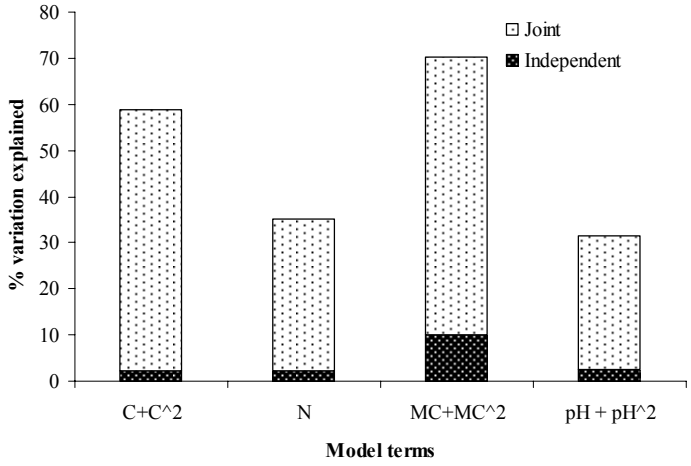


Figure 3. Partitioning observed variation in mean Ellenberg indices among explanatory variables selected for inclusion in the best minimum adequate model for each index. a. Mean Ellenberg R, b. Mean Ellenberg N. MC = % soil moisture, C = % organic carbon, N = % organic nitrogen.

3.4 Niche model testing

3.4.1 Methods

An indication of the accuracy of each GBMOVE model was gained by comparing model predictions of environmental optima for each higher plant species against their published Ellenberg numbers. The optimum for each modelled species refers to the combination of pH, fertility, soil moisture and canopy height that are most favourable for growth. Hence, these predicted values ought to correlate well with the corresponding Ellenberg N, R and W values (see TU annual report for 2005). Results showed a reasonable degree of fit but in this initial test, optima were missing for a range of species while we suspected other values were prone to error. Following a new analytical rather than numerical derivation of species optima, values were accumulated for all species models and a more comprehensive test could be carried out against observed data.

A subset of niche models were tested against independent species presence data recorded as part of a botanical monitoring program for conservation management of agriculturally managed land in Britain. Monitoring plots were located in six geographically dispersed regions designated as Environmentally Sensitive Areas (ESA). Test data consisted of 244 $4m^2$ quadrats recorded in 1995 or 1996 in a range of species-rich grassland and heathland communities. In each quadrat a full list of all vascular plants and bryophytes was made. In addition, soil was sampled immediately adjacent to each quadrat and analysed to provide representative values of soil variables.

106 plant species occurred in greater than 5 quadrats and were therefore selected for model testing. The performance of niche models for each of these species was tested by a linear logistic regression with observed presence in the ESA dataset as the dependent variable and predicted probability of occurrence from each niche model, as the sole explanatory variable. Two series of predictions were tested. First, niche models were used to predict the occurrence of each of the 106 plant species in each ESA quadrat using the mean unweighted Ellenberg N, R and F values and cover-weighted canopy height derived from the plant species composition of each quadrat. Second, measured soil pH, %C, %N and % soil moisture were used to predict corresponding mean Ellenberg values instead of calculating them directly from each quadrat species list. Average soil moisture content was not measured directly but was inferred from soil texture class (C.N.R. Critchley pers.comm). Comparing the logistic regression results for the two sets of predictions allows an assessment of the impact on model performance of the calibration equations used to predict mean Ellenberg indices from soil properties. For all species the desired outcome was a significant, positive regression coefficient indicating that species presences tended to coincide with high predicted probabilities of occurrence.

3.4.2 Results

Of the 106 plant species models that were tested (9% of the total models produced), results differed depending on whether niche models were solved using mean Ellenberg indices that reflected observed species composition or where mean Ellenberg indices were predicted from observed soil data using calibration equations (Fig 4). In the first situation, 92% of species showed the desired significant positive association between observed presence and predicted probability of occurrence. In the second case this dropped to 63%. These results firstly highlight the reliability of the GBMOVE models when solved using mean Ellenberg values

only. However, the drop in performance when solved using observed soil data, highlight the increase in uncertainty when soil data is translated into mean Ellenberg indices using the calibration equations.

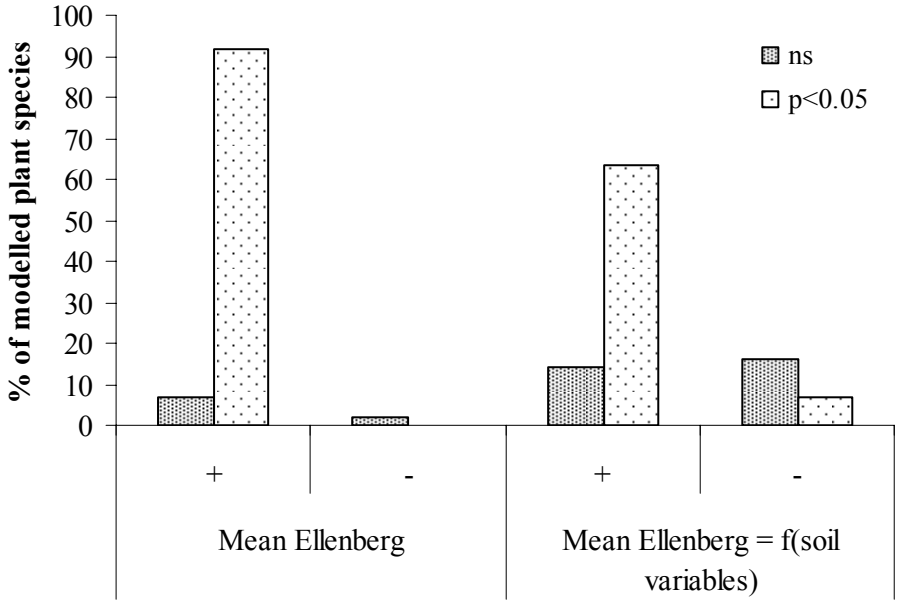


Figure 4. Results of testing niche models for 106 British plant species. Each species was tested by computing a linear logistic regression with observed presence and absence as the response variable in a test dataset of 244 quadrats from British grasslands and heaths, and predicted probability as the only explanatory variable. Predictions were based on solving niche models with either mean Ellenberg values, based on species composition in each quadrat, as the explanatory variables, or by predicting mean Ellenberg values from observed soil pH, soil moisture, % carbon and % nitrogen. In both tests, mean cover-weighted canopy height was also an explanatory variable and was based on observed species composition. Columns labelled '+' indicate positive associations between species presence and predicted occurrence probability.

3.5 Incorporation of climate variables in GBMOVE species models

A second generation of species models has been completed during the current project although not tested. Construction followed the same method as used previously but with the inclusion of min January, maximum July temperatures and precipitation as explanatory variables in each regression equation. Variables were selected using a stepwise algorithm and included all quadratic terms and interactions between climate and all other variables to ensure that differences in abiotic response conditioned by local climate were captured.

Climate data were used at the level of the local 5km sqr containing each geo-referenced quadrat (Table 3). Data were downloaded from the UKCIP web-site as long term averages for the period 1961-1999[6]. While we envisaged that the next stage should be model testing, a

[6] www.metoffice.com/reserach/hadleycentre/obsdata/ukcip

meeting with the sponsors of the MONARCH climate and biodiversity impacts project[7], led us to believe that a trial of other techniques for including climate variables in the niche models would be worthwhile. This work is proposed for the next TU program.

3.6 Modelling species richness and plot size effects

These activities were no longer relevant after the species richness modeling approach was abandoned in favour of predicting habitat suitability indices per species rather than simulating species composition in quadrats of varying size (see TU annual report 2005 for furher details).

4. Species pool selection procedures

A key objective of the Umbrella work program is to generate model-based predictions of change in soil conditions and key plants species for sensitive habitats and soils in British 1km squares. To add realism to species-level predictions an informatics based approach was used to select species-pools for each square. The approach combines two existing databases;

1. Biological Records Centre species lists for each 10km square in Britain.
2. The satellite land cover map for 2000 (LCM2000) which provides Broad Habitat coverage at 25m^2 resolution for the UK.

In order to downscale from the 10km square list to the 1km square, the 10km square species list can be weighted according to the coverage of each Broad Habitat in each 1km square combined with a preference index that gives, for each, species an indication of the Broad Habitat in which each species is most likely to be found. Firstly, species will be selected that have a high preference index for the target habitat. Secondly, species will be further ranked in their likely importance in the surrounding species pool given the coverage of surrounding habitats with which they maybe more or less associated (see below). In essence, the method used to define species-pools for site-based predictions can be simplified for 1km squares. If we can identify where in each 1km square the sensitive habitat is, then exactly the same method could be used with species pools reflecting the Broad Habitat composition in a buffer zone around each habitat area. The simplest method, requiring the least processing would be based solely on the % cover of habitats in each 1km square irrespective of location with in the square.

Our intention was to focus initially on Common Standards Monitoring indicators since changes in these species have an acknowledged impact on the conservation status of the habitat concerned. A worked example for Budworth Common is given below.

[7] www.eci.ox.ac.uk/research/biodiversity/monarch.php

Estimating species pool of potential immigrants based on dispersal traits, species' broad habitat preferences and local broad habitat composition

STEPS:

1. Determine 10km^2 species pool using BRC data or use site species lists where available.
2. Determine broad habitat extent in site (s), in 1500m buffer (b1) around site and in further 1500m buffer (b2). This step uses LCM 2000 and therefore assumes it is accurate at least in the identity and rank abundance of Broad Habitats.
3. Derive an approximate abundance weighting for each species in each zone (s, b1 and b2) using its preference index for each broad habitat as published by Hill & Preston (2003) in combination with the abundance of each broad habitat ie. high indices will reflect a large extent of the broad habitat with which each species is most associated. The index is worked out for species *j* as the sum of the products of multiplying each broad habitat proportion (a value between 0 and 1) by the preference index for the species and the broad habitat. This gives a maximum value of 4. So the index is rescaled by dividing this sum by 4.
4. Multiply the pool abundance index from 3 by the species' dispersal index if available.
5. Sort the CSM indicator table in descending order of the index in 4.
6. Interpret the table on the assumption that each component of the above index is reliable and realistic at the particular site, and that the dispersal ranking is an accurate reflection of real dispersal potential if appropriate vectors are in place.
7. Hence, species with high indices are expected to be most likely to disperse into the monitored patch. Establishment is then hypothesised to be favoured by increased habitat suitability predicted by MAGIC/VSD linked to GBMOVE.

Diagram of index construction for Purple Moor-grass at Budworth common

Broad Habitat composition in site with preference indices in brackets

Bog 3% (4)
Dwarf Shrub Heath 80% (2)
Acid grassland 5% (1)

Sum of proportions * preference

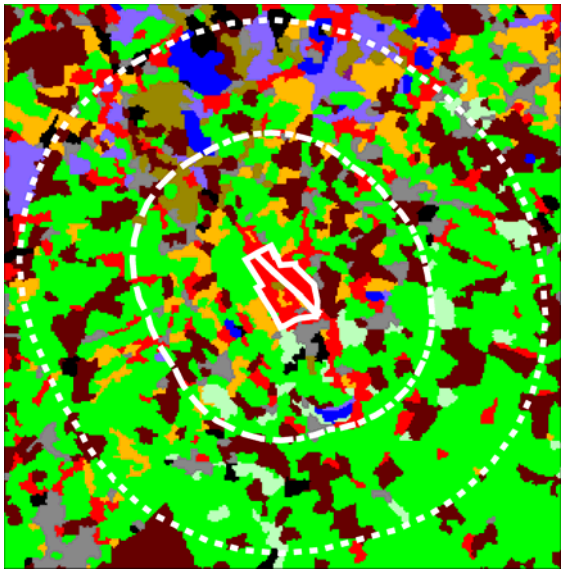
= (0.03 x 4) + (0.8 x 2) + (0.05 x 1)

= 1.77

Index = 1.77/ 4 = 0.44

Then multiply by dispersal index (high = more easily dispersed)

0.44 x 0.43 = **0.19**



Repeat for all CSM indicators NOT recorded in the monitored vegetation, then order table by index in descending order.

5. Testing soil model predictions against observed data from Countryside Survey

In developing our model chain we have sought to emphasize the importance of testing each model component. Only by doing this will we understand the relative size of the uncertainty contributed by each part of the chain and hence where best to target improvements in data quality (input parameters and training data) or in the mechanistic basis of the model. The components of the model chain are illustrated below along with their testing status as carried out during the current and previous TU programs and as part of the parallel Atmospheric N Impacts on Biodiversity project[8].

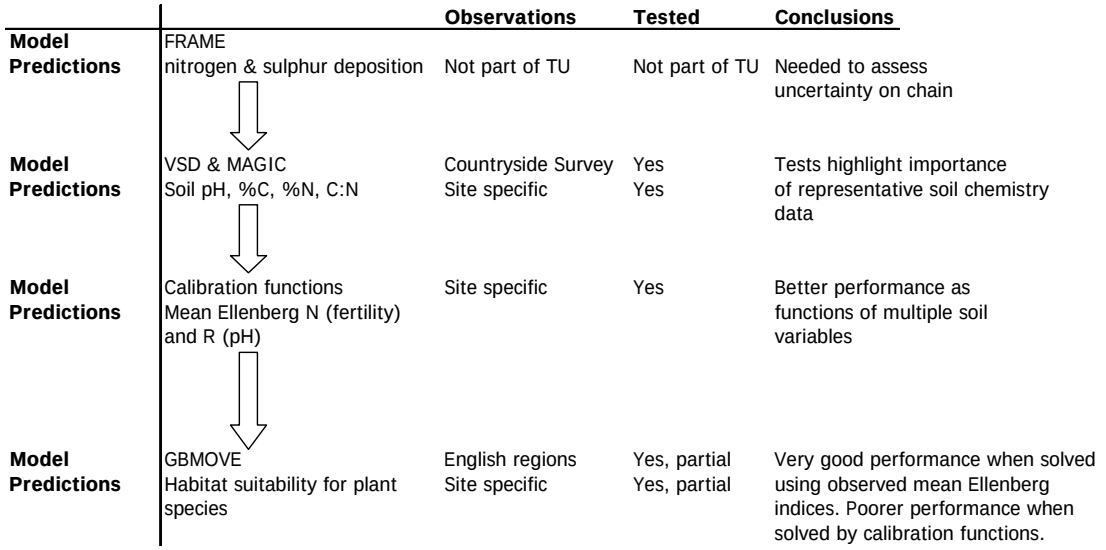


Figure 1. Soil and vegetation model components and their current testing status.

Below we report on the comparison of VSD soil model predictions of pH and C:N ratios with observed data from the 1978 and 1998 Countryside Surveys.

5.1 Methods

5.1.1 Soil modeling

Very Simple Dynamic model (VSD) predictions were made of soil pH and soil C:N ratio for soils sampled in Countryside Survey field plots in 1978 and 1998 (see Black et al 2001 for details of soil sampling procedures). VSD was initialized using National Focal Centre data for critical loads and selectivity coefficients. Soil parameters reflected the soil type within the plot and VSD was run using FRAME N and S deposition scenarios for each wider 5km square (Smith et al 2000, NEGTAP 2001) with upper and lower limits on C:N ratio depending on whether grassland or heath/bog was present in each plot. Predicted values were derived for each year of field survey and could therefore be directly compared with observations from plots in each sample square.

[8] www2.defra.gov.uk/research/project_data/More.asp?I=WC02008&SCOPE=1&M=PSA&V=EP%3A050#Cont

5.1.2 Statistical analysis

Observed soil pH and C:N data were used as the response variable in a General Linear Mixed Model with each 1 km survey square as a random class variable. Scatter plots were visually inspected to identify outliers and distributional problems. Normal errors were assumed and data was not transformed. Explanatory variables were habitat type (acid grassland, heath or bog) and VSD predictions of soil pH and C:N ratio. These were entered as main effects and the interaction term between them also tested to determine whether fit differed between habitat type. All analyses were carried using SAS proc mixed (Littel et al 2000). Degrees of freedom were estimated using the approximation of Satterthwaite (1946).

5.1.3 Results

No significant relationship was seen between modelled soil pH from VSD versus observed soil pH in 1998 CS plots (Table 1). There was no indication that the lack of fit differed between acid grasslands, bogs or heaths as the habitat * VSD interaction was also NS. Predictably modeled pH differed markedly between habitat types.

Table 1. Mixed model results for a comparison of observed soil pH in 1998 versus VSD predictions for 1998.

Effect	Num df	Den df	F value	P
Habitat	2	105	17.14	<0.0001
VSD_pH	1	51.2	0.88	0.3538
Habitat*VSD_pH	2	123	1.79	0.1718

There was no significant relationship between observed *change in soil pH between 1978 and 1998* was compared with modeled temporal change (Table 2). Change in observed pH did not differ between habitat types hence, as expected, the interaction was NS indcating no difference in the relationship between observations and VSD predictions between habitat type.

Table 2. Mixed model results for a comparison of observed soil pH *change between 1978 and 1998* versus VSD predictions of change between 1978 and 1998.

Effect	Num df	Den df	F value	P
Habitat	1	59.4	0	0.9579
VSD_pH_diff	1	29.9	0	0.9815
Habitat*VSD_pH_diff	1	117	0	0.9667

No significant relationship was seen between modeled soil C:N ratio from VSD versus observed soil C:N in 1998 CS plots (Table 3). There was no indication that the lack of fit differed between acid grasslands, bogs or heaths as the habitat * VSD interaction was also NS. The main effect of Habitat was highly significant because this reflects a threshold limit set on C/N to reflect habitat-specific differences in soil organic matter content.

Table 3. Mixed model results for a comparison of observed soil C:N ratio in 1998 versus VSD predictions for 1998.

Effect	Num df	Den df	F value	P
Habitat	2	82	72.55	<0.0001
VSD_C/N	1	112	0.69	0.4084
Habitat*VSD_C/N	2	112	2.81	0.0645

These tests show that substantial improvement in the quality of N and S deposition estimates or/and soil parameter data or/and the process basis of VSD is required. Hence, an imperative is a more detailed analysis of the uncertainty contributed by soil parameters and the FRAME estimates of pollutant deposition (eg. Schouwenberg et al 2001).

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Work package 3:

Dynamic Modelling and Vegetation Response

Task 18:
Modelling of long-term responses of heathlands to N deposition

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Task 18 - Modelling of long-term responses of heathlands to N deposition

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1. Summary

The HEATHSOL-UK model was used to assess the long-term impacts of nitrogen deposition on lowland heaths over the period 1850 to 2050 under different management regimes. New historical deposition scenarios were generated for four regions of the UK by CEH Edinburgh, which included spatial variation within a region.

Our modeling exercise suggests that historical management, rather than historical deposition, has important consequences for lowland heath response to recent increases in nitrogen deposition. We predict that rates of deposition prior to 1950 did not lead to significant increases in nitrogen availability, except in regions where maximum rates of deposition occurred. In contrast, long-term impacts of grazing, cutting and burning on the nitrogen budget of a site were predicted to be more significant in terms of sensitivity of heathland status to increased nitrogen deposition. However, this factor is not considered in most current assessments of the impacts of nitrogen deposition.

Our findings indicate that caution is needed in interpretation of studies which relate current nitrogen deposition to recent ecological change and to biological indicators of possible change, as variation in historical managements between sites may make the 'signal' of nitrogen deposition difficult to detect. The findings relate only to lowland heath, and some consideration needs to be given to their relevance in other habitats for which historical management practices may have differed greatly from those in the more recent past.

2. Policy Relevance

There is an accumulating body of evidence that the ecological impacts of nitrogen deposition are slow and cumulative. This means that models are needed to complement the results of experimental studies. Few such models incorporate management regimes to the level of detail which is needed to understand their long-term impacts on nitrogen budgets. This study suggests that, at least for lowland heaths, management regimes over the last century may be a significant factor in predisposing sites to significant ecological impacts of increased nitrogen deposition in recent decades. This factor, and the sensitivity of model predictions to small-scale variation in nitrogen deposition, suggests that impacts of nitrogen deposition on sites of sensitive habitat within a region may be highly variable. This is very important in interpretation of the evidence of impacts of current rates of nitrogen deposition.

3. Objectives

The original aim of this task was to assess the implications of new long-term experimental data from the field manipulation experiments at heathland and moorland sites for long-term impacts of nitrogen deposition under different management regimes, using the HEATHSOL

model (Terry et al., 2004). However, discussions with the PIs involved in these experiments, Dr. Power (Imperial College) and Dr. Caporn (Manchester Metropolitan University) concluded that there was no indication from the data obtained during the Umbrella research programme to suggest that major changes in model formulation were required. For this reason, since the implications for recovery described by Terry et al. (2004) remain valid, we focused model application on the effects of long-term historical deposition and management on invasion of heathlands by competitive grass species. This is very important to interpretation of studies, including those carried out within this 'Terrestrial Umbrella' contract, seeking spatial correlations between heathland attributes and nitrogen deposition. The objectives were hence:-

- to develop historical deposition and management scenarios for lowland heath
- to assess the importance of recent and historical N deposition in causing changes in the balance of grass species and heather
- to assess the importance of recent and historical management scenarios in causing changes in the balance of grass species and heather.

It should be noted that to meet these ambitious objectives, additional resources from outside the Umbrella contract were used.

4. Methods and results

The model simulations used the HEATHSOL-UK model (Terry et al; 2004) which was originally designed for application to lowland heaths in the Netherlands with sod-cutting as the only management intervention (Heil and Bobbink; 1993). The model has been written in C++ and the model runs on an annual time step and at a spatial scale of one hectare. The mechanisms behind the model predict the outcome of competition for light and nutrients (in this case nitrogen) between a principal heathland dwarf shrub (*Calluna vulgaris*) and a grass (in this case *Deschampsia flexuosa*) under different nitrogen deposition scenarios and management interventions. Plants are divided into compartments of biomass (kg C ha^{-1}) and nitrogen (kg N ha^{-1}) of leaves, flowers, branches and roots for heather and leaves, culms and roots for grass. Biomass for these compartments is calculated based on yearly growth rates which depend on light and nitrogen availability for the whole plant modifying a maximum potential growth rate. Nitrogen concentration in the biomass changes depending on nitrogen availability but remains within specified limits. Total available nitrogen is the sum of the inputs of nitrogen from atmospheric deposition and mineralisation of plant litter.

The model also includes a routine for heather beetle outbreaks the frequency of which are dependent on food quantity, food quality and litter biomass, and are based on a probability of an outbreak occurring once every 15 years (0.06) at low rates of nitrogen deposition. Because of the stochastic behaviour of heather beetle outbreaks the model outputs need to be presented as a mean of a predetermined number of runs. Means of 5, 10, 15 and 20 runs were calculated but 10 runs were found to be sufficiently representative for each scenario, further iterations having negligible effect on the model output mean. All model illustrations are thus based on means of 10 runs.

The model simulations focused on lowland heath, as there is more empirical evidence on interactions between management and N deposition than for moorland systems. The historical simulations were developed for four areas of high importance of lowland heath – Dorset,

Surrey, Cheshire and Breckland. Surrey and Cheshire were included because of the location of the long-term experiments at Thursley Common and Budworth Common. The simulations were based on a 40km grid square using the following centre points:- Cheshire, SJ6060; Breckland, TL8090; Dorset, ST8000; Surrey, SU9030.

The deposition scenarios were developed by Jennifer Mueller at CEH Edinburgh. Ron Smith (CEH Edinburgh) provided the 2000-2004 mean deposition rate of each 5km square within a 40km square, covering the four lowland areas. The 2000-2004 mean provides the baseline from which deposition scenarios were scaled back. The scaling back exercise (trend over 200 years) was applied separately to the total oxidised and total reduced N deposition values. Minimum, maximum and mean deposition scenarios were calculated for each ten years in each area, based on 64 individual 5km square estimates. The partitioning factors for oxidised and reduced N were based on the mean dry and wet deposition of each 40 km square. The maximum NH_3 deposition for one square in the Breckland was 50.3 kg N ha^{-1} yr^{-1}, which is the largest value for the whole of the UK. The large value is most likely due to a very big poultry unit in that square. Since one can reasonably assume that this unit has been present since the 1970s, using this figure for the upper-value historical trend would result in an unrepresentative high dry NH_3 deposition. Thus, this value was considered an outlier, and the second largest value (21.3 kg ha^{-1} yr^{-1}) in the 40 km Breckland square was used instead. Finally, it was assumed, for illustrative purposes only, in these simulations that no change in N deposition occurred after 2005.

The scenarios for maximum total N deposition used in our model simulations are shown in Figure 18.1. Breckland and Surrey have similar intermediate levels of deposition, with Cheshire having the highest and Dorset the lowest levels. Figure 18.1 also illustrates the range of maximum, minimum and mean total N deposition rates for the historical scenarios in the Brecklands which have been the subject of more intense study in this project, as explained below. In all areas, the maximum deposition 5km square was about 50% higher than the mean deposition across the 40km grid square, indicating the potential for significant variation in ecological impacts.

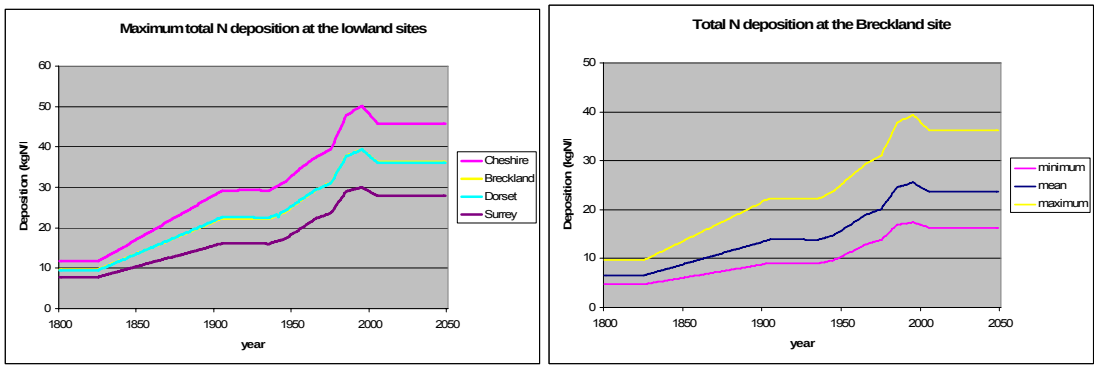

Figure 1. Scenarios of mean N deposition (kg ha^{-1} yr^{-1}) in the four regions (left-hand graph) and range of N deposition scenarios in Breckland (right-hand graph).

The second important component of model data inputs was the definition of historical management regimes. This involved considerable historical research and consultation with UK experts. It is considered that by the mid 18th century, agricultural improvement had

broken the link between heathlands and traditional management as part of the agricultural system and, from this point on, there was no structured management in the way we think of today. However, Webb (pers.comm) suggested that lowland heaths were almost certainly still used for rough grazing and that the vegetation would have been burnt occasionally for scrub clearance and nutritious growth, probably every 10-15 years, but less structured than current burning. The current thinking about managing on a 15-20 year cycle was certainly in place in the 1970s, and may have extended back to the 1950s. Prescribed management (for nature conservation) was in place from the mid 1960s. In the first half of the 1900s there was generally rather limited grazing in the lowlands, and burning or cutting was sporadic. By the 1970s, there was more extensive use of cutting (and burning), to maintain a patchwork of different ages (in many, but by no means all, areas). Grazing is more of a recent re-introduction in lowlands and would still result in fairly low offtakes.

On this basis, given the uncertainty in historical management, we explored the impact of several different historical management regimes. In this report, we focus on the following two basic historical management regimes, which were applied to all four of the lowland regions used in this study, and a grazing regime (see below):

(a) pre-1970: burning or cutting infrequently every 30 years 1970s onwards: cutting at 15 year intervals.
(b) burning every 15 years throughout

In all cases model simulations were started at 1750 to allow the model outputs to stabilize (before N deposition starts to rise) with the proposed pre-1970 management regimes.

Figure 18.2 summarises the outcome of simulations for the four regions with the management regime (a) above, using the 5km square with the highest rate of deposition. It thus indicates the potential for change in each area. To simplify interpretation, the results have been summarized as the ratio of *Calluna* biomass (Cb) to grass biomass (Gb) at the end of each management cycle, expressed as the log of the ratio. Hence a value of 2 or more indicates a continued dominance of *Calluna*, while a value of 0 indicates by definition an equal biomass of *Calluna* and *Deschampsia* at the end of a management cycle. Since *Deschampsia* increases more rapidly than *Calluna* in the early years of the management cycle, this end of management cycle biomass ratio gives only a 'snapshot' of the cover of the two species. For this reason, we interpret values of $\log_{10}(Cb/Gb)$ between 0 and 1 as indicating a potential shift in the competitive balance between *Calluna* and *Deschampsia*.

For all four sites, the plots show the slow decline in *Calluna* dominance with increasing historical N deposition. Comparison with Figure 18.1 demonstrates that this decline in dominance is more rapid at sites with higher levels of deposition. For instance, by year 2000 *Calluna* dominance is being lost at the high deposition Cheshire site (i.e. $\log_{10}(Cb/Gb) < 1$), but still maintained to a considerable degree at the Surrey site, which has the lowest maximum deposition rate. Interestingly, the two intermediate regions, Breckland and Dorset, are both on the verge of beginning to lose *Calluna* dominance to grass (i.e. $\log_{10}(Cb/Gb) < 1$). The change in the biomass ratio is gradual over the modeled period, suggesting that both the pre-1950 and the post-1950 period may be important in accounting for the present balance.

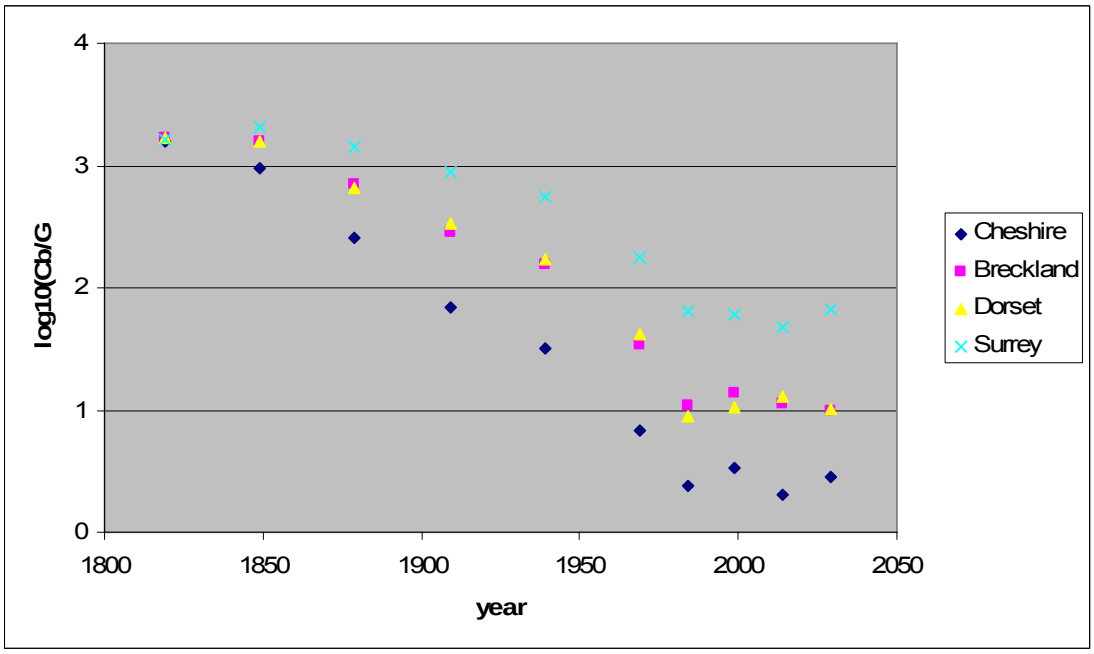

Figure 2. Time-course of the log ratio of *Calluna* and *Deschampsia* biomass over time in the four regions, based on the maximum deposition rate. For further details, see text.

In order to further explore the sensitivity of the predicted change in grass/heather balance to historical and current deposition and management, we focused analysis on the Breckland region. There is some evidence in this region of grass invasion, and we had identified the potential to assess actual historical management in the area, which is known to be different from the other three sites. The Breckland data was used to assess the sensitivity of the current balance between *Calluna* and *Deschampsia* to two major historical factors: deposition and management.

(i) Deposition. For deposition, the importance of variation in both recent deposition and deposition before 1950 was tested and compared for the Breckland region (Figure 18.3). The scenarios used continuous 15 year low temperature burn simulations (scenario (b) above) either with mean deposition rates up to 1950 and maximum, mean or minimum post 1950 deposition rates, or with maximum, mean or minimum deposition rates before 1950 and maximum rates post 1950.

The results firstly show that, using regional mean rates of deposition pre-1950, variation in rates of deposition after 1950 have a large influence on *Calluna* dominance (Figure 18.3a). The mean rate of deposition pre-1950 did not change $\log_{10}$(Cb/Gb), but the increased deposition after 1950 had a significant effect on the balance between *Calluna* and *Deschampsia*. The 5km square with maximum deposition showed a value of $\log_{10}$(Cb/Gb) close to 1 by 2000-2010, indicating a potential for a shift in species composition. In contrast, the simulations with regional mean or minimum values showed little loss of *Calluna* dominance.

The predictions are quite different when historical (pre-1950) deposition is allowed to vary. Reduced deposition rates (regional mean or minimum values) are associated with a steady or increased *Calluna* dominance during pre-1950 period (Figure 18.3b), compared with the

maximum deposition rate, for which there is a small decrease in *Calluna* dominance. However, the variation in deposition rate pre-1950 has almost no effect on the impact of the maximum deposition rate post-1950, with similar values of $\log_{10}$(Cb/Gb) being reached whatever the historical deposition scenario.

Figure 3a

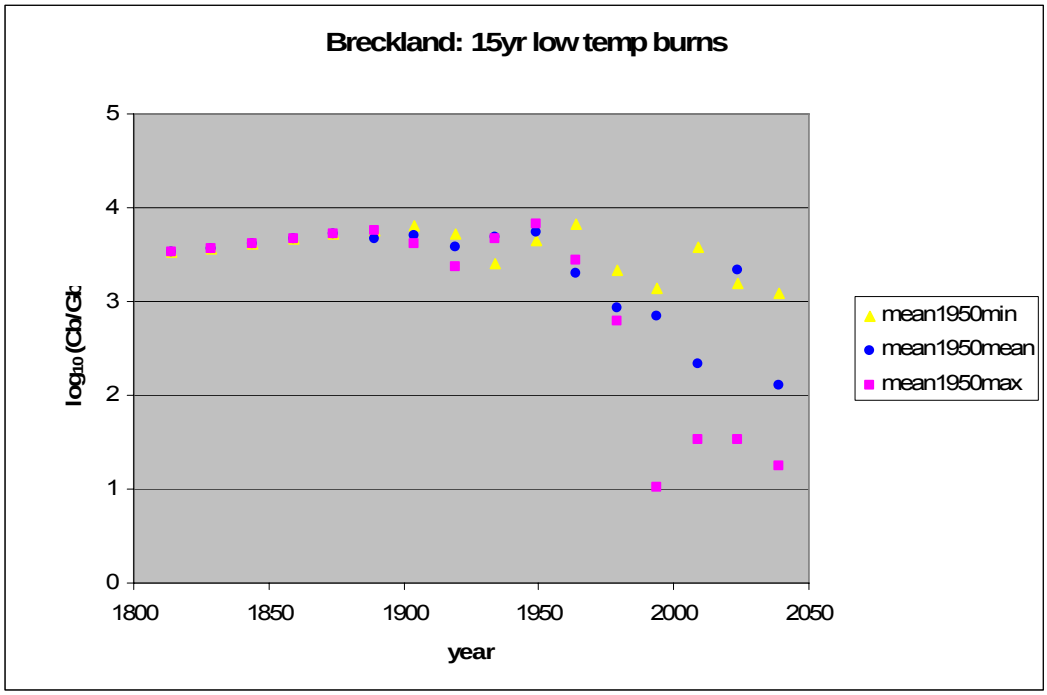

Figure 3b

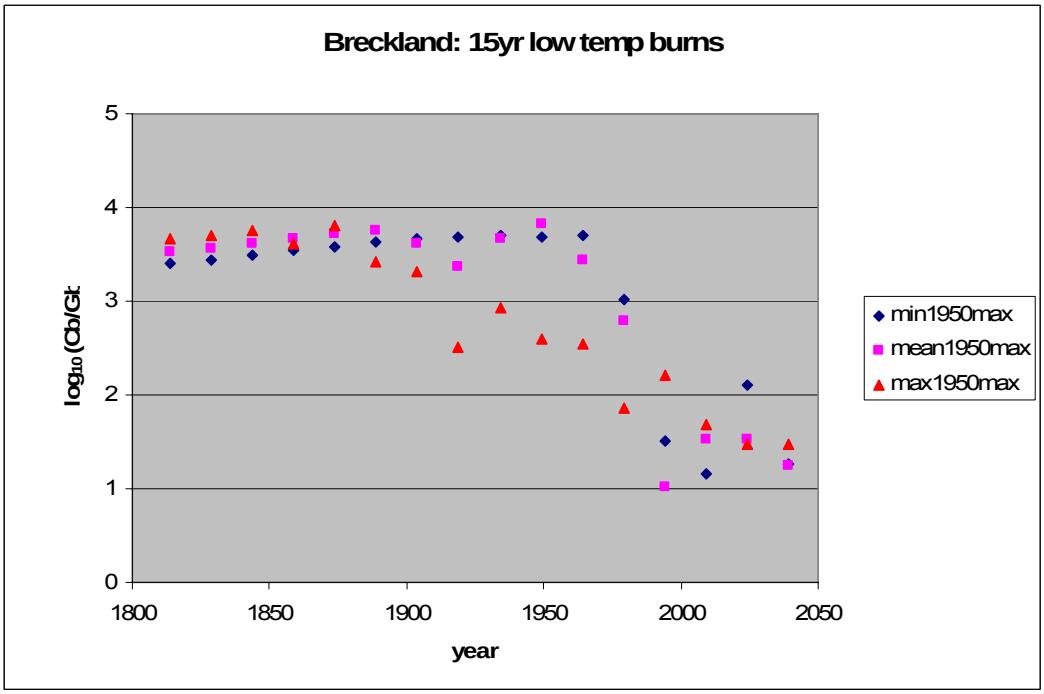

Figure 3. Time-courses of the log ratio of *Calluna* and *Deschampsia* biomass showing the comparison of the effect of variation in recent deposition post 1950 (a) and deposition before 1950 (b) on *Calluna* dominance for the Breckland region.

(ii). Management. Historical evidence suggests that the Brecklands would probably have had a different management history to other lowland areas due to shifting cultivation practices (Webb, pers. comm.). The difficulty for any lowland area is translating exactly what the difference was between ad hoc burning or cutting to support rough grazing and prevent shrub invasion, compared with the more prescribed 15-20 year cycle which has been used since the 1960s. To assess the importance of historical management, we compared three types of historical management:- burning (as used in the scenarios described above), mowing/cutting, and grazing. A basic lowland management of a 30 year cycle of mowing/cutting was used to approximate to ad hoc managements pre-1970s, followed by a 15 year cycle.

Research by Sibbett (pers. comm.) suggests that rabbits were the primary grazers of heaths in Breckland in the 1800s and early 1900s. Britton (pers. comm.) also brought to our attention a book by E. Pickworth Farrow (1925) which has many photographs of Breckland heaths in the 1920s showing that the amount of rabbit grazing had a striking impact on heathland vegetation. There may have been some sheep activity, but generally not year round, with densities of about 1-2 per ha. Numbers of rabbits were high. Trist (1979) quotes: "*At Elveden in 1900 the gamekeepers took 83,000 rabbits which rose to 128,000 in 1921. By 1940 the catch from a reduced population was 42,000 and to 19,500 by 1952*". Therefore, the area must have been heavily grazed. He goes on to say: "*In 1972 the Elveden flock stood at 900, whereas in 1927 it was almost 2000. Rabbits are now very much reduced compared with the former population*". So by the 1970s rabbits had almost gone (mainly due to myxomatosis in the mid 1950s) and there were not many sheep; Watt (1974) reported that Foxhole Heath (a chalk grassland site) was intensively grazed by rabbits during 1938 but that when the rabbits died out in 1954 there was only sporadic light grazing by sheep and cattle.

Based on this information, we adopted a fairly intensive rabbit grazing management pre-1950, followed by 15 year low temp burns. A literature search provided some evidence of grass offtakes for rabbit grazing pre-1950. Documentary evidence for typical high density rabbit populations points to a value around 50-60 rabbits ha^{-1} (Sheaill, 1979): McKillop *et al*. (1996) reported an estimated 1% loss in yield per rabbit per hectare for winter wheat. Therefore, assuming a maximal rabbit population of about 60 ha^{-1}, grass offtakes would be in the region of 60% per hectare per year. No data for levels of *Calluna* grazing have been found in the literature, so we assumed that selective grazing by rabbits is similar to that for sheep with *Calluna* offtake half that of grass offtake (Terry *et al*., 2004). Rabbit grazing at high intensities did not have a large effect on *Calluna* dominance although absolute productivity levels were reduced. However, low grazing levels produce a slight increase in grass competitiveness post 1960 during the burn managements. This is as expected, since as long as rabbits selectively graze the grasses, then they will maintain *Calluna* dominance.

Figure 18.4 summarises the effects on *Calluna* dominance of the major different historical management regimes described in this study for the Breckland region in combination with a maximum nitrogen deposition scenario. This figure strongly suggests that, in contrast to historical deposition scenarios, historical management has a significant effect in modifying the impact of the increase in N deposition after 1950. In summary the decline in *Calluna* dominance with increasing nitrogen deposition is reduced when the frequency of managements is increased (in this case 15 year low temperature burns) compared to the original basic 30/15 year low intensity mowing (also shown in Figure 18.4). Pre-1950 rabbit grazing at moderate levels (40%:20% offtake of *Calluna*:*Deschampsia*) maintains *Calluna* dominance to an even greater degree than the 15 year burns, but the lower rabbit grazing intensity (20%:10%) has a smaller effect.

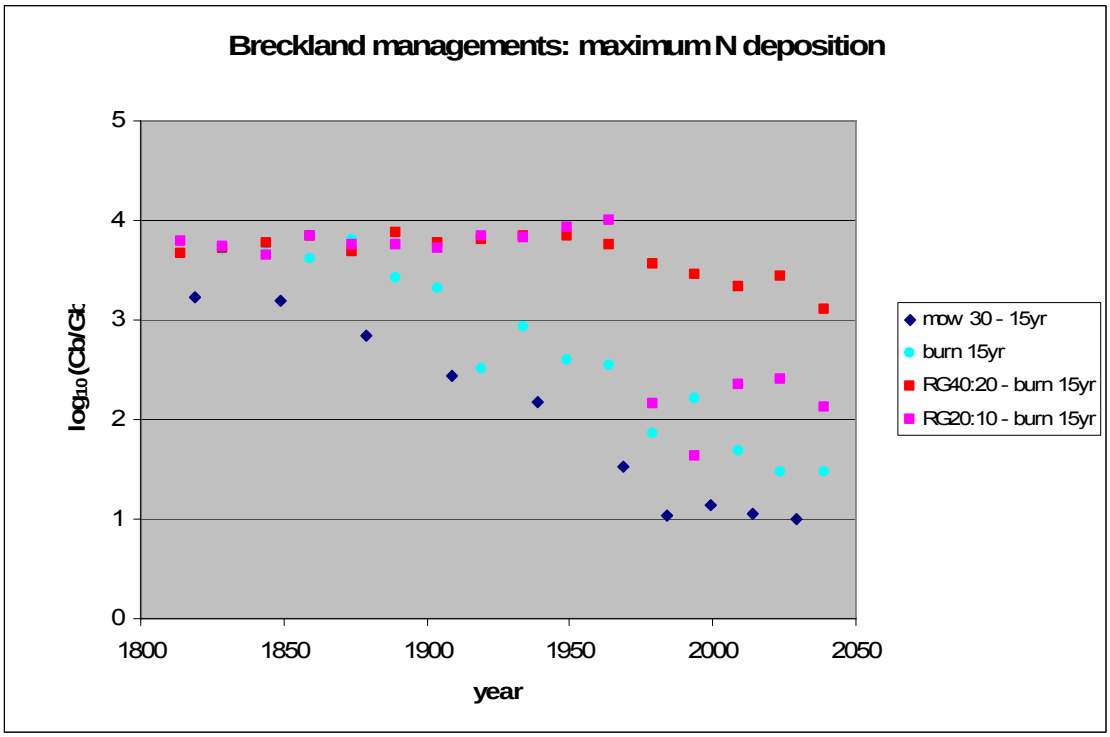

Figure 4. Time-courses of the log ratio of *Calluna* and *Deschampsia* biomass showing the effects on *Calluna* dominance of the major historical management regimes in the Breckland region. RG40:20 and RG20:10 refer to rabbit grazing intensities -see text for details.

5. Discussion and implications

The results of this exercise should not be interpreted as predictions of the likelihood of changes in species composition in lowland heath caused by nitrogen deposition. There are uncertainties in model formulation and parameterisation (see Terry *et al*., 2004), although the model has been tested successfully against experimental data for the Budworth Common and Thursley Common sites (Power *et al*., 2004). Furthermore, there are many uncertainties in the historical deposition data, while historical management regimes are quite speculative. Therefore, it is the broad pattern of the model results that is important for interpretation.

The finding that historical management, and not historical deposition, is important in explaining the impacts of more recent increases in nitrogen deposition, is most likely due to their impacts on nitrogen budgets. It is likely that rates of deposition prior to 1950 did not lead to significant increases in nitrogen availability, except in those grid squares with maximum rates of deposition. In contrast, the long-term impact of grazing, cutting and burning on the nitrogen status of a site is likely to be more significant, and explain the sensitivity to increased nitrogen deposition in the recent past. This is a factor which is simply not considered in most current assessments of the impacts of nitrogen deposition.

These findings have several implications for the interpretation of studies which relate current nitrogen deposition to current ecological change and to biological indicators of possible change. Such studies often show a large variation between sites which makes the 'signal' of nitrogen deposition difficult to detect. Our results suggest that this may be a problem in studies where (a) variation in current rates of deposition do not correlate well with variation

in cumulative rates of deposition over the last 30-50 years but no earlier and (b) there is substantial variation in historical management between sites. Clearly, these findings relate only to lowland heath, and some consideration needs to be given to their relevance in other habitats for which historical management practices may have differed greatly from those in the more recent past.

6. Possible future work

Work using the HEATHSOL model will not continue under the new 'Terrestrial Umbrella' contract. Nevertheless, this type of model is very important in allowing interactions between management and nitrogen deposition to be assessed over the long time-scales which are appropriate in terms of these major ecological impacts of nitrogen deposition. The role of historical management in 'pre-disposing' sites to impacts of more recent increases in nitrogen deposition has barely been explored, and it is important to investigate whether there is evidence for these types of effects on the ground, perhaps in an area such as Breckland where good records of historical management exist.

The work under this contract has focused on biomass of the two major competing species as an indicator of significant ecological change. However, the HEATHSOL-UK model generates information on a range of other response parameters, including changes in soil chemistry and nutrient balance that are central to interpreting the model outcomes. Since regional surveys of bio-indicators of effects of nitrogen deposition include such variables, further analysis of the model outputs would be valuable in terms of interpretation of the results reported here, and in interpreting the findings of such surveys. A peer-reviewed paper based on the results is in preparation.

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Annex 1 - Task 14 Pristine peat pH

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Annex 1 - Task 14 Pristine peat pH

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On a global scale the temperate peatlands cover about 3.5 million km^2 of land and contain approximately 455 Gt of carbon, this is approximately equivalent to the entire carbon store in all life within the biosphere. In addition they also represent about 25% of all the soil carbon on earth. Bogs are a sink for atmospheric carbon and their ability to absorb carbon (sequestration) has been estimated to account for approximately 12% of current human emissions. They are characteristically low in biodiversity but their flora and fauna are distinctive and unique to this habitat. Peatlands/bogs are an important resource for scientific research, including the study of past environments and climate change (Moore, 2002).

Threats to peatlands come from direct human exploitation in the form of peat harvesting for energy and horticulture, and drainage for forestry and agriculture. Peatland drainage will result in losses of carbon to the atmosphere, adding to the greenhouse effect. Fragmentation of the habitat is not an important consideration because bogs are by there very nature 'island' habitats. The effects of acidification are less certain. Moore (2002) states that acidification by aerial pollution may be a local problem close to sources, but the habitat is naturally acidic and so should not be severely affected. Inputs of other aerial pollutants however particularly nitrogen could have widespread impact on bogs, altering their vegetation and enhancing their productivity.

Climate change is likely to be the biggest threat to temperate bogs. Higher temperatures if accompanied by raised carbon dioxide levels and enhanced nitrogen deposition will enhance productivity but will also result in faster decomposition rates. Combining these two opposing factors and adding in future rainfall patterns will determine future peat formation. If summers become warmer and drier and winters become milder and wetter, the summer drought could cause peat loss and bog contraction. Excess decomposition may result in bogs becoming a net carbon source and a positive feedback to global warming. Methane and nitrous oxide emissions would add to the greenhouse effect but oxidation of methane and low nitrous oxide emission may mean this source is not significant. Tree invasion of bogs because of the summer drought could locally lead to an increase in water loss through transpiration, and higher heat absorption through albedo change, enhancing the drying effect on the bog surface. The review by Moore concludes by saying that the future of northern bogs is likely to be determined by the quantity and pattern of future precipitation.

Most of the world's peatland (86.4%) are found in the temperate zone with 99.4% of these lying in the northern hemisphere. Most of these peatlands lie in the cool temperate zone, between latitudes 50°N and 70°N in Russia, Fennoscandia and north-west Europe, Canada and the United States of America (Moore, 2002). In the southern hemisphere temperate peatlands are located in the lowlands of western Chile (Pisano, 1983), there are also some in Tasmania, south-eastern Australia, and New Zealand (Campbell, 1983). Peatlands can also be found in isolated patches on some islands such as the Falkland Islands.

Historically the classification of peatlands has been the source of much debate. Traditionally they have been split into fens and bogs or rheotrophic (flow-fed) and ombrotrophic (rain-fed)

mires (Moore & Bellamy, 1973). The term 'bog' is normally reserved for peatlands which receive all their water supply directly from rainfall with no contribution from drainage water. However, it is often confusingly used to describe acidic, nutrient poor peatlands that receive some through-flow of ground water. Alternatively bogs and fens can be separated on the basis of pH and subdivided by substratum fertility. Splitting by pH reflects the bi-modal ph-frequency distribution in mire waters and the region of most rapid floristic change (Wheeler & Proctor, 2000). The acidity of wetlands depends on the balance between metallic cations and strong acid anions, which is dependant on the composition of the source waters and their capacity to buffer acidity naturally produced by plants, from acid rain or produced by other means. The review by Wheeler and Proctor through consideration of the gradients present within mire ecosystems; ombrotrophic-minerotrophic, low fertility-high fertility and water table for example, suggest roughly the divisions illustrated in figure 1.

Human beings have had a generally negative impact on peatlands, the most important ones being drainage for forestry and agriculture, cutting and harvesting of peat for fuel, atmospheric pollution, and recreation and education. In order to evaluate if the critical pH suggested by Cresser is still valid we need to find peatlands which have been exposed to as few of these pressures as possible. Ombrotrophic peatlands (bogs) obtain their nutrient inputs from the atmosphere and from precipitation. *Sphagnum* mosses are dependent on nitrate deposition from the atmosphere for the supply of this element (Proctor, 1995). *Sphagnum* mosses are not nutrient demanding and so are easily out competed by more competitive species if supply of nutrients is plentiful. The increase in the grass *Molinia caerulea* in blanket mires to the west of Britain has been correlated with industrial growth and nitrogenous air pollution (Chambers, Mauquoy & Todd, 1999).

Sulphur dioxide in the atmosphere has been directly implicated in the decline of *Sphagnum* mosses in the southern Pennines. It has been shown through field experiments that *Sphagnum* mosses are killed off by SO_2 in solution (Ferguson & Lee, 1979). The effect is worse in oligotrophic conditions; plants with a more abundant mineral supply may rely on transition metals to detoxify the impact of the dissolved bisulphite ions, oxidizing them to sulphate (Lee, 1998). Increased sulphur content and decreased pH in precipitation can influence nitrogen availability in the acrotelm. Nitrate is immobilised and NH_4^+ is released from the acrotelm and surface litter layer of ombrotrophic mires with increased H, N and S inputs in precipitation (Sanger, Billett & Cresser, 1996).

Efforts to reduce sulphur and nitrogen emissions have been successful with reductions of approximately 25% in sulphur dioxide emissions since 1990 to 2003. NO_x emissions have also been reduced by almost 50%; ammonia however has not seen any decrease and this could be of concern to the health of UK mires. Overall these reductions in atmospheric pollutants are good news for UK mires, however on a more global scale air pollution including acidification, sulphur enhancement and nitrate enrichment, is likely to continue and will influence the development and survival of cool temperate bogs. Acidification will cause fens to become more oligotrophic, rich fens may also become acidified even though they are buffered by ground water calcium carbonate. Ombrotrophic mires are already acidic and unless sulphur deposition becomes extremely heavy are not likely to be affected (Moore, 2002). Sphagna are unlikely to suffer damage as long as the soil surface pH remains above 3 (Lee, 1998).

The aim of this review is to establish whether the critical pH of 4.4, the point at which noticeable damage to peat ecosystems is visible, set by Cresser (date) is still valid or if it

needs refining. Considering the review so far it would seem sensible to have separate critical pHs for different peat types and that a baseline of 4.4 is too simplistic. The natural pH of fens and bogs, or rheotrophic and ombrotrophic mires depending on what terminology you use, is different. All peatlands whether ombrotrophic *Sphagnum* bogs, pH approximately 4, or minerotrophic swamps and fens, pH 6-8, contain dissolved acids from the decomposition of plant materials: carbonic acid and organic acids. Minerals then combine with the acids produced through decay to regulate the pH. The pH of peatland waters is therefore controlled by acid production from the decomposition of organic matter and supply of bases from ground and surface waters (Shotyk, 1988).

If the acids produced through the decomposition of organic matter can be neutralised by dissolved bases it is possible to have a relatively high pH (6-8.5). The physical and chemical composition of the neutralising waters depends on the local rocks and sediments. As peat accumulates over time, the thickness increases and groundwaters exhibit less influence. The end result is an ombrotrophic *Sphagnum* bog. The bog receives little outside neutralising water and so the bog waters exhibit low pH values. In this review we are asking how low the pH of ombrotrophic *Sphagnum* bog can go before significant damage occurs.

Evaluating the published literature to determine this question is difficult because of problems of interpretation. Measurements of peat pH, peat pore water pH, and peatland surface water pH show different pH values. Peatlands also show large variations in pH with depth. *Sphagnum* bogs from around the world generally show surface water pH values of around 4, see Shotyk (1988) for references. In contrast to Wheeler and Proctor (2000), Shotyk splits peatland types into the ranges shown in table 1. The pHs from moist peat, raw (unfiltered) water, dried peat or salt solution can all be different so comparison of like with like is important.

Table 1. The range in pH of peatland types (from (Sjors, 1950)

Peatland Type	pH Range
Bog	3.7 - 4.2
Extreme poor fen	3.8 - 5.0
Transitional poor fen	4.8 - 5.7
Intermediate fen	5.2 - 6.4
Transitional rich fen	5.8 - 7
Extreme rich fen	7 - 8.4

There are vertical and horizontal variations in pH, Rhodes (1933) reported values of 4.0 in a *Sphagnum* bog, and pH values of 6.8-7.4 in the swamp margin. A peat profile from St. Fergus Moss (NK052536) showed variation of 3.85 at the surface down to 4.48 at seven metres. There are geological, hydrological, pedological and climatic factors also to consider when interpreting peatland pHs. Peats underlain by non-calcareous sediments had lower pH values than sites underlain by calcareous sediments. The chemical composition of groundwaters on pH has already been mentioned but consideration of the size, shape, and slope of the drainage basin, the height of the local groundwater table, the hydraulic conductivity of the local mineral soils and sediments and peats needs considering. The lowest pHs have been shown to be areas with the thickest peat. In addition climatic conditions such as wet and dry periods and seasonality effect peat pH. Comparison between wet and dry periods has shown a decrease in pH of 2 units during the dry period as compared to the wet. Analysis of peat at the Environmental Change Network site at Moor House-Upper Teesdale National Nature

Reserve has shown trends in the mobility of dissolved organic carbon in peat soils related to drought-induced acidification (Clark et al., 2005); another consequence of either natural or anthropogenic acidification.

The review by Shotyk concluded that, published pH values cannot be interpreted quantitatively in terms of H^+ activity. This is because the true pH is difficult to determine due to colloidal organic materials, volatile acids (CO_2 and organic acids), and oxidizable acids and bases. In addition the situation is further complicated by horizontal and vertical variation, seasonal differences and the different methods used to determine peatland pH. Published values of peat pHs need to be critically evaluated according to all these factors.

(Tahvanainen & Tuomaala, 2003) assessed the differences in mire water pH obtained by different sampling methods. The study assessed variation according to the poor-rich gradient, the water-table-depth gradient, time of day, aeration of samples, and different methods of obtaining samples. Considerable diurnal and fine-scale vertical and horizontal variation of pH was found in samples by depressing from bryophyte capitula (what does this mean?) or directly from open water surfaces. A diurnal pattern of lower pH at night and higher during the day was found, with variation of between 0.5 and 1.0 pH unit. Open surface water samples showed a wide horizontal variation but pipe-well samples were less variable in both aerated and unaerated samples. The results also confirm a bimodality of pH covering the transition from rich to poor fen. The paper also suggests a standard protocol for water sampling in mires which could be considered in studies of mire waters.

Soils in Scotland were surveyed from the pre-war years to the late eighties. The areas surveyed were primarily lowland cultivated soils with the aim being to assess the areas suitable for conversion to intensive agriculture. Where peats were found these were sampled, their vegetation described, the peatland composition determined and chemical analysis undertaken. The memoirs produced from this survey contain the data from these analyses and provide useful information when building a picture of 'pristine' peatland pH in the UK. When interpreting the data from the memoirs it was important to try and take into account the history of these sites i.e. previous management regimes. The text of the memoirs was studied and an expert decision was made as to whether the sites visited were pristine or not. The issues already discussed re. sampling strategy also needs to be considered.

Figure 1 shows the pH of the surface horizon of ombrotrophic peat bogs. These have been subdivided into disturbed bogs (blue), unknown bogs (red) and pristine bogs (yellow) according to site history. The average pH of each of these groups are 3.6, 3.9 and 3.6 respectively. Of the peats that were described the majority were dominated by *Sphagnum*, the rest were a mix of *Sphagnum, Eriophorum* and *Calluna*. The *Sphagnum* peats had an average pH of 3.6. From the data from the soil memoirs it hasn't been possible to distinguish pH differences as a result of vegetation type but previous management re. cutting and ditching has had a negligible effect.

Figure 2 shows the location of these bogs and their pH. A comparison of the location of the bogs with the acidic deposition in that area is illustrated in figure 3. Looking at the surface pHs of the bogs and the deposition around them there isn't a strong correlation to suggest acidic deposition has caused significant acidification. For example the sites in the south west are in an area with historically high acid deposition yet they do not show extensive acidification. The sites in the far north east, are the most pristine having received little or no acidification but there are still some sites with surface pH as low as 3.4. A more

comprehensive study of peatland pH in Scotland was undertaken by Skiba *et al* in 1989. Peat was analysed from 123 sites, most peat samples were taken from the top 15 cm of the peat horizon, some from the top 20 cm and three from the top 25 cm. The peats were dried, sieved and stored before analysis in calcium chloride. The drying and storing process will have altered the true pH making the samples more acid. However the paper still gives a good general outline of peat pH in the mid to late 80s in Scotland. The sites chosen for analysis are unlikely to be pristine as it was not the purpose of the study to evaluate pristine peat pH. But to look at whether acid deposition could explain some of the modelled changes in pH. The authors found peats with highest acidity ($pH(CaCl_2) \leq 3.0$) and with lowest base saturation ($\leq 10\%$) are found mainly where deposited acidty was greater than 0.8 kg H^+ ha^{-1} yr^{-1} (Skiba et al., 1989).

The National Soils Resources Institute in 2003 under DEFRA contract assessed changes in organic carbon content of non-agricultural soils in England and Wales. Originally in the National Soils Inventory (NSI) 1281 points were sampled, under this contract (SP0521) 580 of the original sample points were revisited. Grouped by major soil group, 72 of those were classified as peat soils. If an average of all positions is taken no significant change in pH was found between 1980 and 2003, pH increased by 0.1 from 3.9 to 4.0. If grouped by land use 34 of the 580 were classified as bogs, average pH in 1980 was 4.3 and this showed no change to 2003. pH was determined in a 1:2.5 suspension of soil in water to match the original analysis in 1980. Using this dataset there has been no change in peat pH in England and Wales between 1980 and 2003 (Bradley, 2003).

Scientists in Fennoscandia have studied peatlands extensively because it is a common soil type in this area. In Finland during 1975-2000 the Geological Survey of Finland (GTK) investigated 1.7 million ha of the 5.1 million ha of the geological mires in Finland. The study covered about one million study points and 12000 single mire basins. The final report assesses the peat reserves of Finland and their exploitability. Figure 5 shows the pH of peatland across Finland. The pH's shown are averages through the profile, the mire waters being measured directly with pH probe pushed directly into the moist peat sample. Peats are more acid in the south and west and this would seem to coincide with the more pristine areas of Finnish peatland. However the differences are better explained by differences in the underlying geology and vegetation. In the north there is much more calcium and magnesium in the bedrock and glacial till which is the main soil type than in south and central Finland. There are also great differences in the vegetation and peat types. In the south and central Finland there is mainly Sphagnum bogs (S.fuscum, S. angustifolium, S. magellanicum, S. ledum) which have a naturally low pH but in northern Finland the dominant vegetation and cover type is fens with herbs. In central Lapland this consists mainly of plants such as Bryum, Scorpidium, Drepanocladus, Sphagnum warnstorfii, sedge (Carex) species.

The Finnish definition of pristine in the report is based around ditching and drainage. If a sampling point was closer than 30 cm to a ditch then the point was not classed as pristine, the age of the ditch was not deemed a significant consideration. The Finns think that the influence of ditching or drying affects only the surface peat i.e. the top 50 cm. In close to neutral peat (pH 5-6) the ditching will lower the pH towards the acid by one pH unit. But in very acid peats (pH 3-4) there is no significant change in peat by ditching.

Peatlands acidify naturally as they develop. The natural pH of a peatland depends on what stage it is in its sucession towards a rain fed ombrotrophic bog. The Proctor (2000) paper gives a good outline of what vegetation should be expected at different pHs and fertilities,

this is summarised in figure 1. Reviewing data from the UK and Scandinavia has shown peat bogs surviving at pHs below 4.4 suggesting the acid deposition load that would give rise to an effective rain pH value of 4.4 is too high. It depends on which species you are looking to preserve or what species should or would have been present in a pristine world.

FIGURES

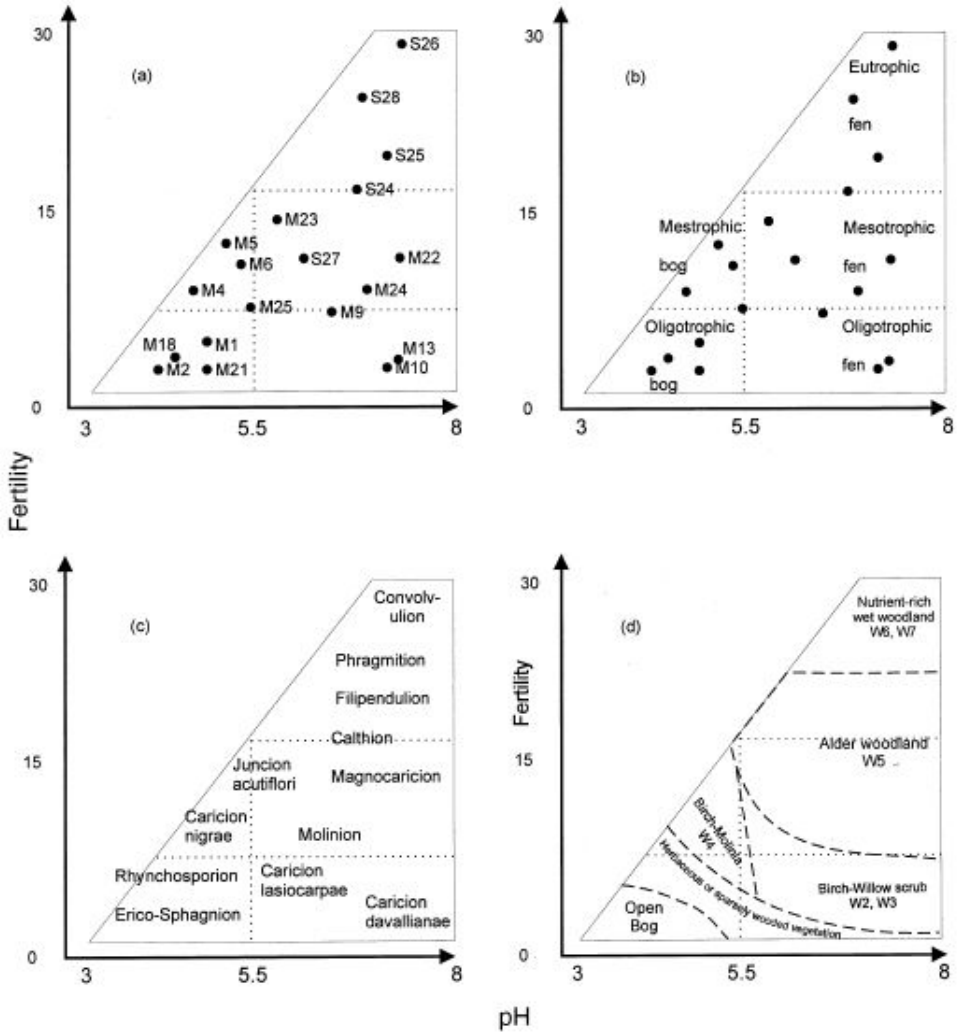

Figure 1. Variation of wetland in Britain in relation to pH and substratum fertility (estimated phytometrically by growth of *Phalaris arundinacea* seedlings on substratum samples; y-axis values are mg plant^{-1} from the data Shaw & Wheeler 1991). (a) Approximate position of NVC plant community types of Rodwell (1991a, b, 1994). (b) Proposed subdivision of mires into broad pH and trophic status categories; category boundaries are approximate. (c) Schematic arrangement of the main phytosociological alliances of mires in relation to the proposed pH and trophic status categories. (d) Schematic arrangement of the main categories of self-maintaining mire vegetation in Britain that might be expected to occur in the absence of human disturbance, plotted in relation to water pH and substratum fertility. The separating boundaries are tentative, and in general the categories will intergrade. The units shown represent the presumed climax or pro-climax state, excluding woody phases that may precede these in succession. The extent of non-wooded vegetation in this diagram will depend on factors including climate, topographical situation and water level.

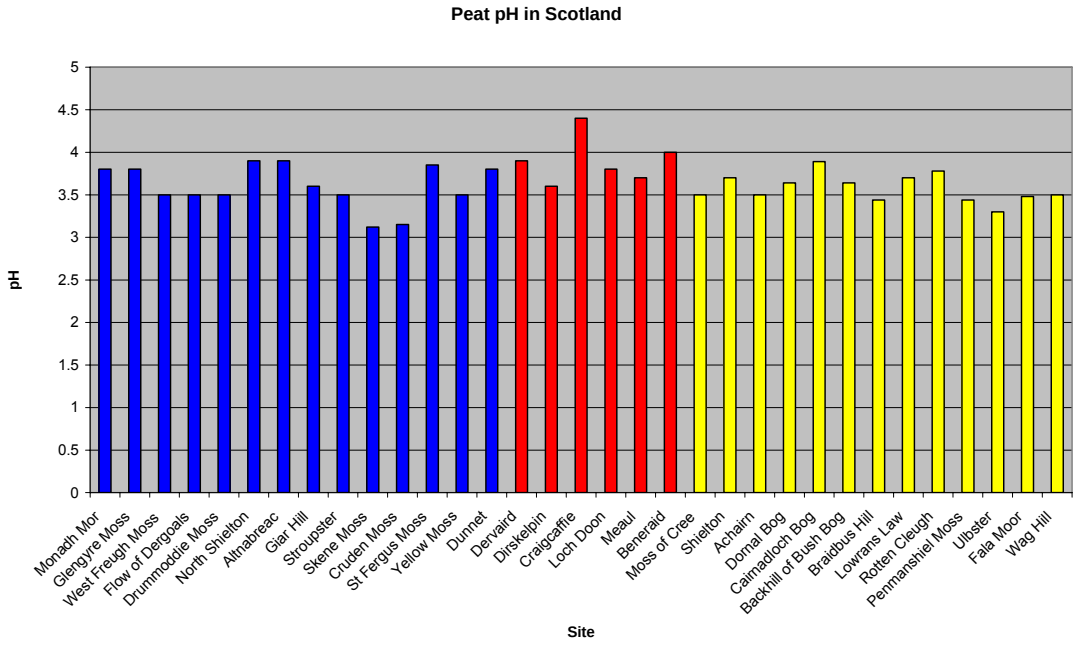

Figure 2. Peat pH in Scotland, data from Soil Survey of Scotland Memoirs.

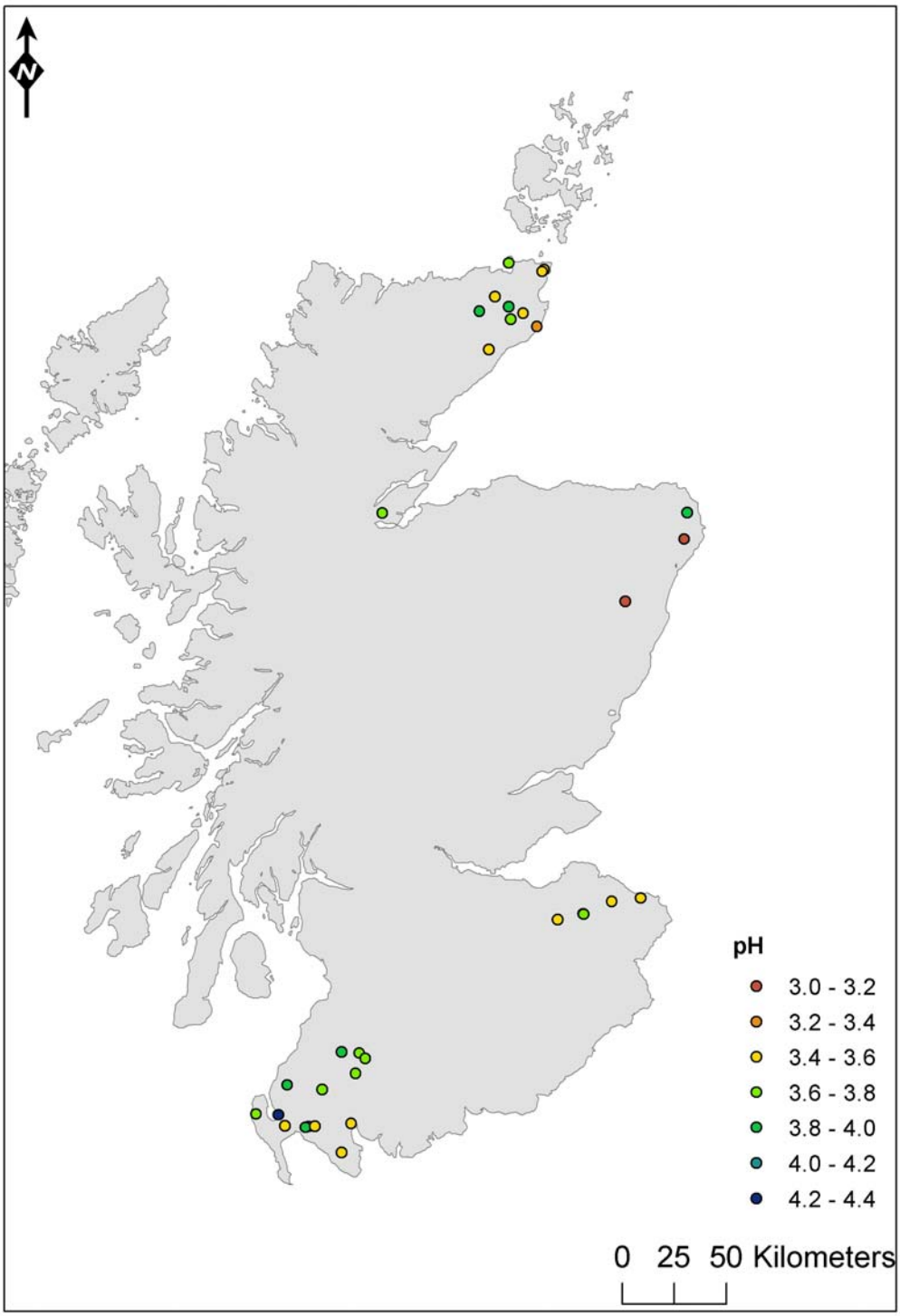

Figure 3. Surface horizon peat pH in Scotland, data from Soil Survey of Scotland Memoirs

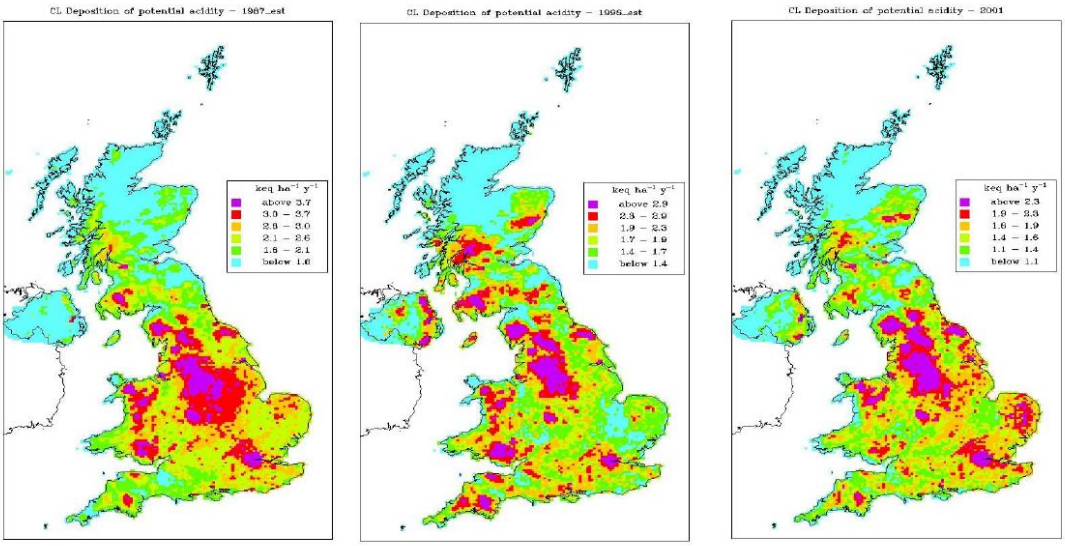

Figure 4. Total acidifying inputs 1987 (estimated), 1996 (estimated) and 2001 (Fowler et al., 2004)

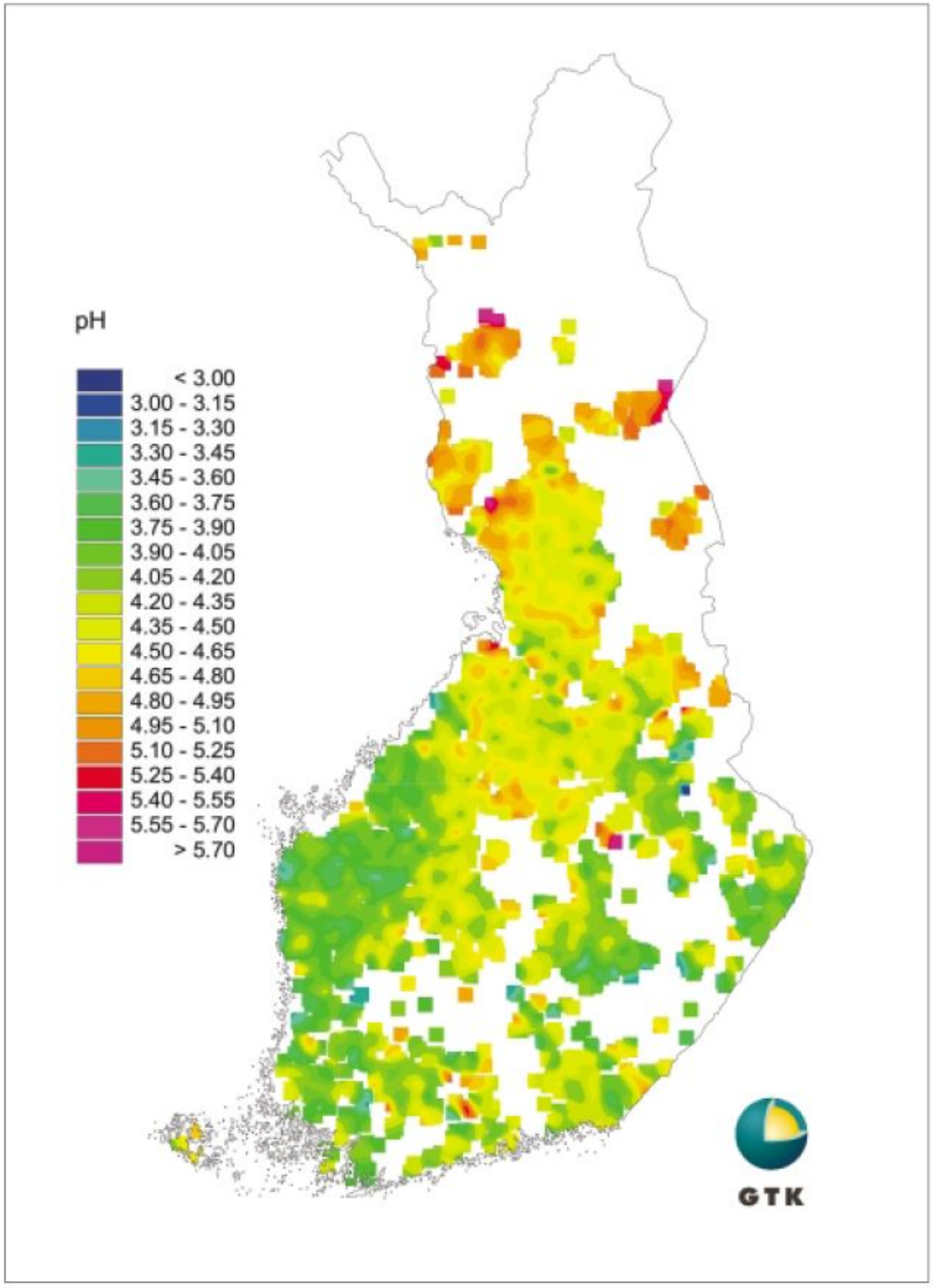

Figure 5. Regional variation of peat pH in Finland (white area on the map – no research data in GSF's files)

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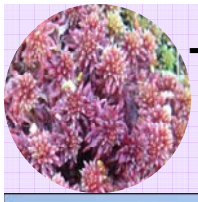
ANNEX 2

Mini syntheses of impacts of acid
and nitrogen deposition on blanket and raised bogs



The impacts of acid and nitrogen deposition on:

Blanket and raised bogs

The UK currently has 1.5 million hectares of blanket and raised bog, including 6000 hectares of lowland raised bog, a significant proportion of the worlds' resource. Bogs are an ancient, peat based ecosystem, representing thousands of years of organic matter accumulation, predominantly where rainfall is highest, in the west and north. Bogs support specialised plant communities, adapted and restricted to nutrient limited conditions that are sustained by the wet, often anoxic, acidic conditions which restrict decomposition. Bogs are priority habitats for nature conservation, valued for their specialised plant and bird communities and their ability to act as a sink for carbon. Large areas of bog were drained last century for agriculture and forestry, today increased atmospheric nitrogen (N) deposition, climate change, grazing and trampling threaten their existence.

The distribution of inputs of acidity and nitrogen across the UK

Energy production through combustion of fossil fuels results in the emission of nitrogen oxides (NO_x) and sulphur dioxide (SO_2) into the atmosphere. Food production also emits pollutants: ammonia (NH_3) from farm animal units, and both ammonia (NH_3) and nitrous oxide (N_2O) from intensive fertiliser use. These are transported in the atmosphere affecting air quality and rainfall chemistry across the UK. This pollutant deposition (Figure 1) has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to N enrichment of semi-natural areas. Reductions in emissions due to policy control measures have resulted in lower quantities of sulphur and nitrogen oxides falling on different habitats but, due to increases in emissions from shipping, recovery has not been as fast as hoped for. Ammonia emissions increased sharply from the 1950s to 2000 and currently remain at these peak levels.

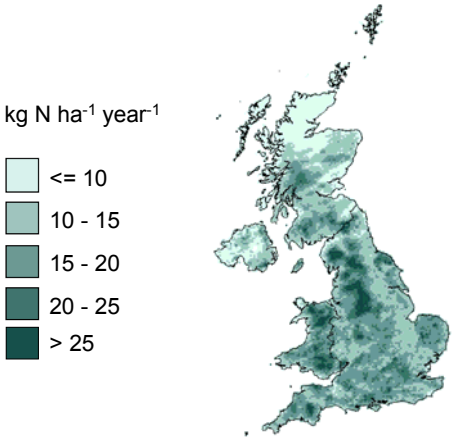


Fig 1a. Nitrogen (NOx + NHx) deposition measured 2003-2005

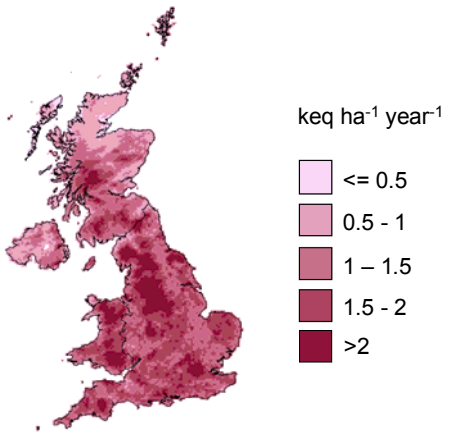


Fig 1b. Total acid deposition (S + NOx + NHx) measured 2003-2005

Participants in the **UK** **R**esearch on **E**utrophication and **A**cidification of **T**errestrial **E**cosystems programme include: Centre for Ecology & Hydrology, ADAS Pwllpeiran, Forest Research, Imperial College London, Macaulay Institute, Manchester Metropolitan University, University of Sheffield and the University of York

www.bangor.ceh.ac.uk/terrestrial-umbrella

Evidence of acidification and N-enrichment effects at the national scale

There are various sources of information which indicate vegetation, soils and waters have been affected by acidic and N deposition. A review of the evidence for the UK was brought together by the National Expert Group on Transboundary Air Pollution (NEGTAP) (http://www.nbu.ac.uk/negtap/home.html). The evidence for N enrichment of vegetation includes two national monitoring programmes – the Countryside Survey and the New Plant Atlas for the UK – which identified shifts in species composition towards more nutrient-demanding species in the latter half of the 20th century (Preston *et al*. 2002, Haines-Young *et al*. 2003) (e.g. Figure 2).

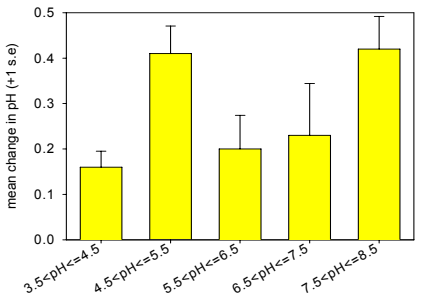


Figure 3. The increase in soil pH (0-15cm) from 1978-1998/9 at 767 sites across Great Britain (NEGTAP 2001)

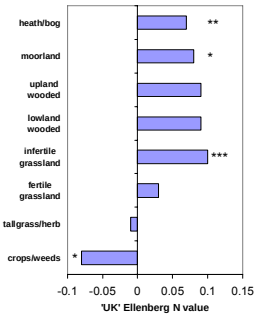


*Figure 2. The change in UK fertility score for different habitats between 1990 and 1998 taken from the most recent UK Countryside Survey (redrawn from Haines-Young et al., 2003). A low value is associated with low fertility and a high value with high fertility. Statistically significant changes are marked as * P<0.05, ** P<0.01, *** P<0.001.*

Acidification of soils and waters recorded in some areas during the 20th century are now being reversed, reflecting the success of emission policies to reduce levels of acid deposition in the environment (e.g. Figure 3). There are still areas at risk, however, due to increases in sulphur emissions from shipping.

Why does air pollution affect our soils, vegetation and waters?

Although rainfall is naturally acidic, additional acidity either introduced directly by sulphur dioxide and nitrogen oxides or formed during the breakdown and uptake of ammonia has affected waters, soils and vegetation in the UK. The pH of lakes and rivers fell during the last century, in turn affecting populations of fish, invertebrates and water plant communities. Soils also became more acidic, affecting organic matter breakdown and soil nutrient balance. Soil acidification increases the solubility of some elements such as aluminium in the soil solution, which can be toxic to plant roots at high concentrations. Pollutants are also deposited to vegetation directly as gases, aerosols and in fogs and mists, and can cause direct damage to plants at high concentrations.

Emissions of nitrogen oxides and ammonia can lead to N enrichment (eutrophication). These problems can result in a loss of biodiversity in sensitive ecosystems because N-loving species benefit at the expense of other species of conservation interest that contribute so much to the character of semi-natural habitats. This happens due to nutrient imbalances, increased susceptibility to climatic stress and higher levels of insect or fungal damage which all affect the balance of competition between species.

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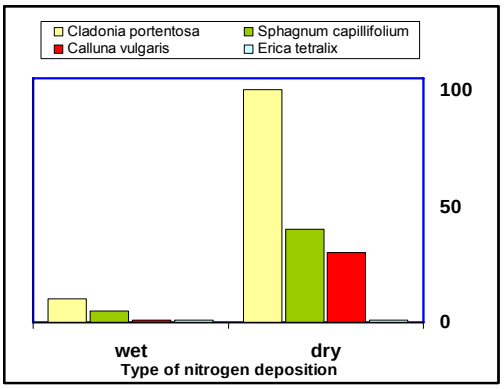
Current evidence for air pollution effects on bogs

Research on the effects of N deposition on bog ecosystems is limited, although there are many studies on responses of individual species of *Sphagnum* moss. In addition there is a large amount of field evidence from intensively farmed regions of the Netherlands and Denmark showing that high concentrations of gaseous ammonia and ammonium in precipitation have led to the loss of *Sphagnum* mosses, keystone species for the sustainability of acid bogs.

In the UK, the majority of bogs occur in the north-west where N deposition is predominately associated with the large amounts of precipitation. An experiment was established in 2002 at Whim bog in the Scottish Borders to assess the effects of wet deposited N at loadings comparable to those received by UK bogs (majority < 20 kg N ha^{-1} yr^{-1}), and compare them with those of released gaseous ammonia, simulating an agricultural point source.

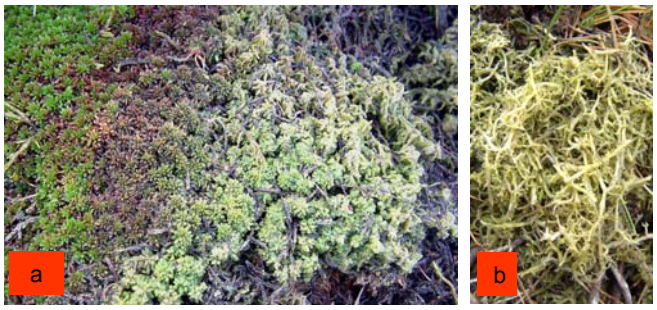
Aerial photograph of Whim bog nitrogen manipulation experimental site.

Additional N (~20 kg N ha^{-1} yr^{-1}) provided in precipitation for 5 years, on top of a background deposition of 8 kg N ha^{-1} yr^{-1}, has not yet caused significant changes in species composition, or species loss, among any of the functional groups, namely lichens, mosses, *Sphagnum* mosses, sedges or dwarf shrubs (such as bilberry, heather). There is evidence of increased growth and N accumulation, although these responses are species specific. Doubling this amount of N suggests that in the long-term, at current levels of deposition, hummock forming *Sphagnum* species such as *Sphagnum capillifolium* may be lost from those bogs in high N deposition areas. Ammonia at concentrations providing a similar N dose (~20 kg ha^{-1} yr^{-1}) is much more toxic, and affects more of the key species.



Percent damage caused by five years of added nitrogen, ca. 20 kg N ha^{-1} yr^{-1} in precipitation or as ammonia gas.

Nitrogen effects on bogs appear to be strongly influenced by climatic conditions which can predispose plants to detrimental effects of N and *vice versa*. Increased injury from spring frosts, winter desiccation, summer droughts and fungal pathogens have all been observed.



a) Stages of Sphagnum death and b) dead lichen (Cladonia portentosa) in response to increased nitrogen deposition.

However, invading species appear to be less of a problem than in other habitats, as mosses quickly colonise the exposed litter layer.

In the Whim experiment, N retention has been high to date, but increases in mineral N are measurable in the rooting zone which will have implications for a variety of microbial driven processes, such as greenhouse gas emissions and carbon sequestration.

Over this relatively short time scale changes in emissions of methane and nitrous oxide have been small, but longer term data are needed.

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How will climate change and management affect the impacts of air pollution?

Redistribution of rainfall, in both time and space, could represent a real threat to bogs where regional climates are predicted to become drier and warmer. Research suggests that such changes will greatly exacerbate the detrimental effects of N deposition on bogs, and *vice versa*. Both *Sphagnum* species and *Calluna* (heather) are highly sensitive to desiccation stress. Climate change may also accelerate the breakdown of peat and lead to shifts in community ecology, with the ultimate loss of bog ecosystems. Management intervention by increasing the amount of water draining into bogs, and damming up drainage outflows could help offset a falling water table. Trampling causes long-term physical damage to bog vegetation which can be addressed through sensitive management.

Drain blocking using pile and sheeting dams on the RSPB reserve at Forsinaird, Sutherland. Photograph by D. Norrie.

On bogs where *Racomitrium lanuginosum*, woolly hair moss, is dominant, reducing the grazing pressure may help offset anthropogenic effects. Restricting the type of plants grown in close proximity to bogs, particularly tree species may help control invasive species.

UK actions being taken to help reduce air pollution

Protocols under the UNECE Convention on Long Range Transboundary Air Pollution (CLRTAP), have already led to substantial emissions reductions for sulphur dioxide and nitrogen oxides. As a result, acid deposition in the UK has declined by approximately 50% over the past 12 years, mainly due to reductions in sulphur emissions. Under the latest CLRTAP agreement (the Protocol to Abate Acidification, Eutrophication and Ground-level Ozone) UNECE parties have agreed more stringent emission ceilings for SO_2 and NO_x as well as the first emission ceilings for NH_3, to be met from 2010. A major driver for agreement of these ceilings was the aim to reduce exceedance of critical loads for acidification and eutrophication across Europe. Critical loads are defined as the amount of acidity or nutrient N deposited on an ecosystem that, if exceeded, could lead to damage of that ecosystem. Critical loads are improved and refined as new data on ecosystem impacts become available. A recent update of UK critical loads has been undertaken and the report is available at: www.critloads.ceh.ac.uk .

Further Information

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ANNEX 3

Mini syntheses of impacts of acid
and nitrogen deposition on lowland heath

The impacts of acid and nitrogen deposition on:

Lowland Heath

The UK currently has approximately 58,000 ha of lowland heath, representing 20% of the international total for this habitat. Lowland heathland occurs at altitudes of less than 300 m and is typically associated with nutrient-poor, often sandy soils. Both wet and dry heathlands are priority habitats for nature conservation and, in addition to supporting a diverse flora and fauna, heathlands also have a high amenity value. Lowland heaths are managed systems, requiring the regular removal of nutrients to maintain nutrient-poor conditions. However, since the late 19th Century, a decline in traditional management practices, together with changes in land use and increasing urbanisation, have resulted in the loss of large areas of heathland. More recently, elevated deposition of nitrogen (N) is thought to have contributed to widespread heathland decline throughout NW Europe.

The distribution of inputs of acidity and nitrogen across the UK

Energy production through the combustion of fossil fuels results in the emission of nitrogen oxides (NO_x) and sulphur dioxide (SO_2) into the atmosphere. Food production also causes pollutant emissions: ammonia (NH_3) from farm animal units and both ammonia (NH_3) and nitrous oxide (N_2O) from intensive fertiliser use. These pollutants are transported in the atmosphere affecting air quality and rainfall chemistry across the UK. This has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to N enrichment of semi-natural areas. Reductions in emissions due to policy control measures have resulted in lower quantities of sulphur and nitrogen oxides falling on different habitats but, due to increases in emissions from shipping, recovery has not been as fast as hoped for. Ammonia emissions increased sharply from the 1950s to 2000 and currently remain at these peak levels.

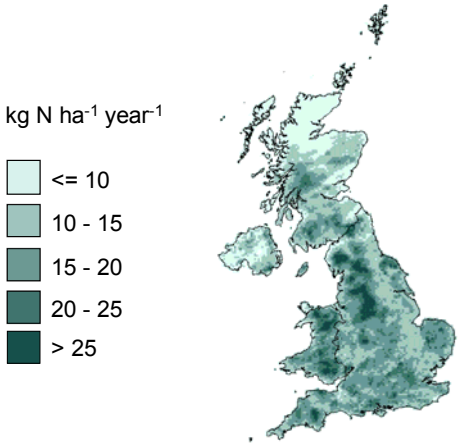

Fig 1a. Nitrogen (NOx + NHx) deposition measured 2003-2005

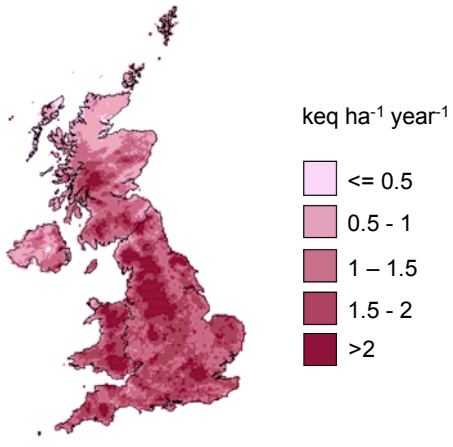

Fig 1b. Total acid deposition (S + NOx + NHx) measured 2003-2005

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Evidence of acidification and N-enrichment effects at the national scale

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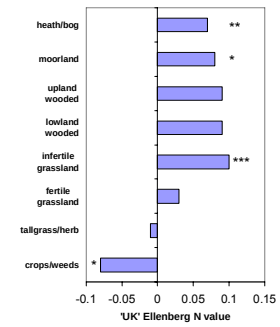

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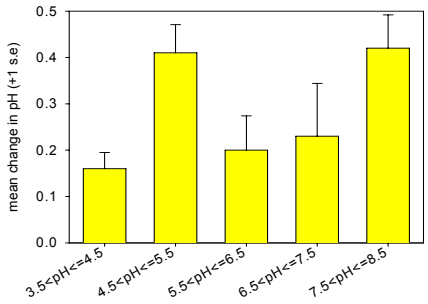

Figure 3. The increase in soil pH (0-15cm) from 1978-1998/9 at 767 sites across Great Britain (NEGTAP 2001)

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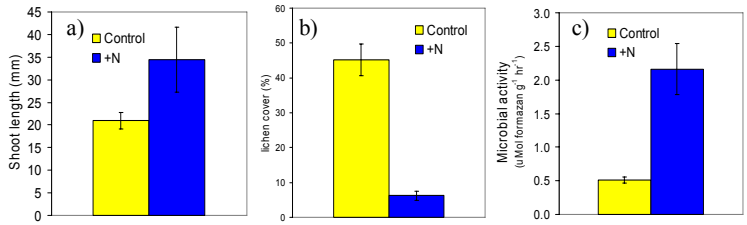
Current evidence for air pollution effects on lowland heaths

Research in the Netherlands, Denmark and the UK has demonstrated a variety of effects of N deposition, and to a lesser extent acidity, on the growth, chemistry and species composition of heathland vegetation and on nutrient cycling within heathland ecosystems. Results from long term manipulation experiments at lowland heaths in both north-west and southern England have shown that even relatively small increases in N inputs result in increased growth of heather (*Calluna vulgaris*) as well as earlier bud burst and higher foliar N concentrations. These changes are frequently associated with an increase in sensitivity to drought and frost as well as faster growth of the heather beetle, an insect which feeds exclusively on heather and which can be responsible for wide scale damage to vegetation.

Experimental nitrogen addition at Thursley Common NNR

In the Netherlands, elevated rates of N deposition have been responsible for increased dominance of the heathland grasses wavy hair grass (*Deschampsia flexuosa*) and purple moor grass (*Molinea caerulea*), at the expense of heather (Aerts & Heil, 1993). A recent nationwide survey has shown that the occurrence of the dominant heathland dwarf shrub species, and heathland habitat, have decreased significantly in the UK during the past 20 years (CS2000). However, whilst there is some evidence linking N deposition with a change in heathland community composition in Britain, field experiments have shown only a short-lived increase in grass species following prolonged periods of N addition (Wilson, 2004; Barker *et al*., 2004).



Effects of nitrogen addition (30 kg ha^{-1} yr^{-1}) on a) Calluna *growth, b) lichen % cover and c) soil microbial activity at Thursley Common heathland NNR (Surrey).*

Lichens and mosses are considered to be particularly sensitive to N deposition; heathland manipulation studies have shown negative effects of N on the performance and diversity of moss and lichen communities (Haworth, 2005; Carroll *et al.*, 1999).

Measuring the fate of added nitrogen in heathland soils

Although N deposition is the major pollutant issue for lowland heathland, acidity and high concentrations of ammonium ions have also been shown to have direct toxic effects on sensitive herbaceous species in Dutch heathlands (de Graaf *et al*., 1998). Whilst the effects of N deposition are most clearly seen above-ground, changes below-ground are also observed, with important consequences for plant nutrient and water relations. Typical below-ground responses include a reduction in *Calluna* root growth, a build up of soil N stores and changes in the rate of nutrient cycling.

The evidence to date indicates that a large proportion of N inputs is immobilised in the soil microbial biomass. Changes in microbial community composition and an increase in microbial activity are associated with accelerated rates of decomposition and faster nutrient cycling, factors which may favour the growth of more nutrient-demanding species, such as grasses, over slower growing dwarf shrubs, like heather. Current and future legislation will reduce emissions of nitrogenous pollutants, but little is known about the ability of semi-natural ecosystems to recover from the effects of eutrophication. Ongoing work at Thursley Common, a lowland heath in Surrey indicates that recovery will be a slow process, with the effects of earlier N inputs persisting for many years after additions cease.

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How will climate change and management affect the impacts of air pollution?

Air pollution is not the only driver of ecosystem change; climate change is likely to have detrimental effects on heathland vegetation and alter nutrient cycling. Research has shown that N addition increases the sensitivity of heather to drought; climate change may result in even greater levels of drought injury, particularly in combination with elevated N deposition. Another important issue is that temperature is frequently a limiting factor for insect and microbial performance; warmer temperatures are likely to result in increased herbivory and faster nutrient cycling. Lowland dry heathlands typically occur on well drained, sandy soils, with limited water holding capacity. Predicted changes in summer rainfall and temperature may result in a greater frequency of uncontrolled summer fires, with detrimental effects on soil structure and seed bank, as well as heathland fauna.

Habitat management, in the form of controlled burning, turf cutting, mowing or grazing, is used as a tool to maintain low nutrient levels in lowland heaths. Recent results from both experiments and modelling studies indicate that frequent, intensive management (for example turf cutting or mowing with litter removal) is needed to retain nutrient-limited conditions at many heathland sites under current levels of N deposition.

Management burn of an experimental heathland plot

UK actions being taken to help reduce air pollution

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ANNEX 4

Mini syntheses of impacts of acid
and nitrogen deposition on upland heath

The impacts of acid and nitrogen deposition on:

Upland Heath

Upland heath covers extensive areas of moorland between enclosed farmland and montane regions on nutrient poor, acidic surface soils (usually peats or peaty podzols) in the north and west of the UK. A product traditionally of burning and grazing management, the plant community is dominated by heather (*Calluna vulgaris*) and other shrubs including bilberry (*Vaccinium myrtillus*). The economically and ecologically important vegetation provides grazing for grouse, sheep and deer, and habitat for diverse bryophytes, lichens, invertebrates and raptor birds. Large areas were lost in the 20th century due to changing land use, over-grazing and management neglect. In parts of Britain, high levels of sulphur dioxide and acid rain have damaged the soils and vegetation of upland heath and related moorland ecosystems. Recently, nitrogen (N) deposition has become recognised as an additional threat to the structure and function of these nutrient-poor communities.

The distribution of inputs of acidity and nitrogen across the UK

Energy production through the combustion of fossil fuels results in the emission of nitrogen oxides (NO_x) and sulphur dioxide (SO_2) into the atmosphere. Food production also causes pollutant emissions: ammonia (NH_3) from farm animal units and both ammonia (NH_3) and nitrous oxide (N_2O) from intensive fertiliser use. These pollutants are transported in the atmosphere affecting air quality and rainfall chemistry across the UK. This has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to N enrichment of semi-natural areas. Reductions in emissions due to policy control measures have resulted in lower quantities of sulphur and nitrogen oxides falling on different habitats but, due to increases in emissions from shipping, recovery has not been as fast as hoped for. Ammonia emissions increased sharply from the 1950s to 2000 and currently remain at these peak levels.

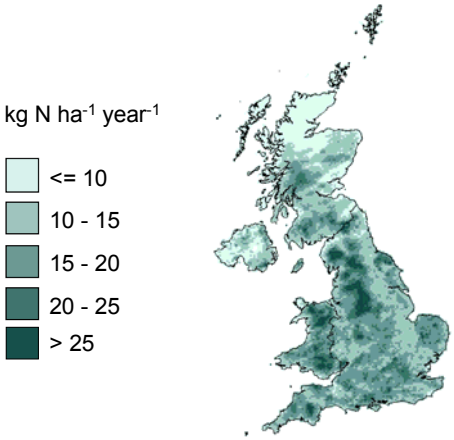

Fig 1a. Nitrogen (NOx + NHx) deposition measured 2003-2005

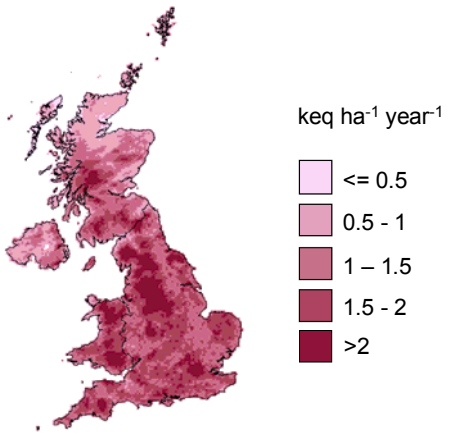

Fig 1b. Total acid deposition (S + NOx + NHx) measured 2003-2005

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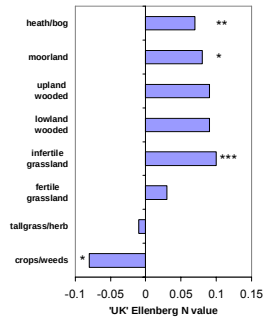

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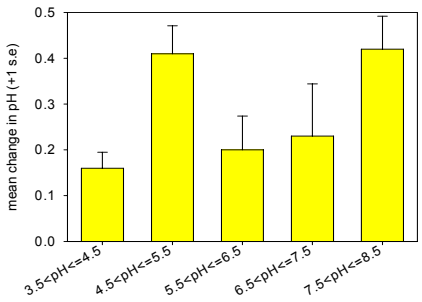

Figure 3. The increase in soil pH (0-15cm) from 1978-1998/9 at 767 sites across Great Britain (NEGTAP 2001)

Acidification of soils and waters recorded in some areas during the 20th century are now being reversed, reflecting the success of emission policies to reduce levels of acid deposition in the environment (e.g. Figure 3). There are still areas at risk, however, due to increases in sulphur emissions from shipping.

Why does air pollution affect our soils, vegetation and waters?

Although rainfall is naturally acidic, additional acidity either introduced directly by sulphur dioxide and nitrogen oxides or formed during the breakdown and uptake of ammonia has affected waters, soils and vegetation in the UK. The pH of lakes and rivers fell during the last century, in turn affecting populations of fish, invertebrates and water plant communities. Soils also became more acidic, affecting organic matter breakdown and soil nutrient balance. Soil acidification increases the solubility of some elements such as aluminium in the soil solution, which can be toxic to plant roots at high concentrations. Pollutants are also deposited to vegetation directly as gases, aerosols and in fogs and mists, and can cause direct damage to plants at high concentrations.

Emissions of nitrogen oxides and ammonia can lead to N enrichment (eutrophication). These problems can result in a loss of biodiversity in sensitive ecosystems because N-loving species benefit at the expense of other species of conservation interest that contribute so much to the character of semi-natural habitats. This happens due to nutrient imbalances, increased susceptibility to climatic stress and higher levels of insect or fungal damage which all affect the balance of competition between species.

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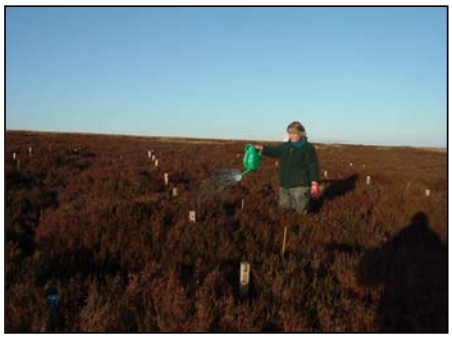
Current evidence for air pollution effects on upland heaths

The influence of current acid and nitrogen deposition on upland heath and similar moorland ecosystems is seen against the background of lasting effects of historical pollution since the Industrial Revolution. In the worst-hit areas, such as the Southern Pennines, the increased soil acidity, loss of base cations and inputs of metals have left a long-term legacy which appears to constrain present-day ecological recovery. In the 1980s, it was predicted that, following a reduction in emissions of sulphur dioxide, ecological improvements in polluted moorlands would be slowed due to the influence of continuing high rates of N deposition; research in recent years supports this view.



Experimental nitrogen addition on a Welsh moor

Lichens and bryophytes provide the main above-ground biodiversity in many upland heath communities and a long term N addition experiment near Ruabon in North Wales found these plants to be highly sensitive to N inputs (Pilkington *et al.*, 2007). Acidification has occurred over extensive areas of the UK. Indeed a link between peat acidification and rainfall acidity across Scotland was established in the 1980s (Skiba *et al.*, 1989). Soil acidification slows the growth rate of plants, some more than others, so communities change. Research has shown that decomposition of plant litter slows down in acidified soils, substantially so at heavily polluted sites, and this is not sustainable in the longer term. Peat acidification can reduce the population of enchytraeid worms, important to the first stage of litter decomposition, and can reduce substantially the colonization of heather roots by beneficial mycorrhizal fungi.

Nitrogen deposited to heather moorland initially accumulates in the mosses and higher plants and then builds up in the surface soil layers where the slow rates of decomposition and mineralization of organic latter limit the leakage of N into waters. The amount of N leaching as a proportion of deposition has been found to be much greater in some areas, like the Southern Pennines, than in others (Curtis *et al.*, 2004). Detailed research in this area indicates that microbial retention of the high N inputs is reduced in heavily polluted and frequently burnt moorland soils (Cresser *et al.*, 2004). The reduction in lower plant cover in the moorlands of the most polluted regions also means that this vegetation, especially the moss layer, is less able to absorb and retain atmospheric N inputs.

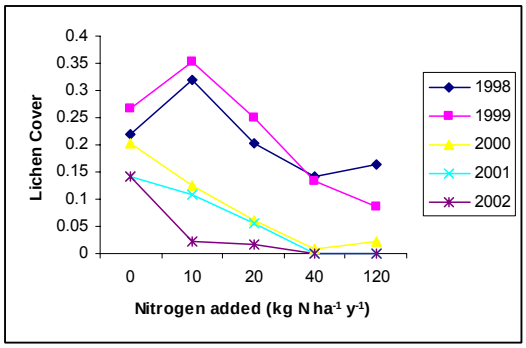


Figure 4. The changing effect of ammonium nitrate additions on lichen cover in the first 5 years of an experimental on a Welsh moor

Heather is the dominant and most valued species of moorlands but evidence for direct effects of N pollution on it is equivocal. Some pot experiments found strong, adverse effects on roots and mycorrhizas, yet studies around the country found very good heather growth in well managed moors even in regions with the highest acid and N deposition (Milne *et al.*, 2002). In the Ruabon field experiment, monthly additions of ammonium nitrate solutions since 1989 have not substantially changed the condition of the dominant heather plants, despite major changes to lower plants below the canopy (Figure 4.) and to soil chemistry.

Damage to the dominant shrubs may result indirectly from environmental stresses such as freezing, exposed conditions or insect (heather beetle) herbivory, both of which can be exacerbated by N deposition, increasing injury in heather shoots.

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How will climate change and management affect the impacts of air pollution?

The combined effects on upland heaths of climate change, rising CO_2 and air pollution are hard to predict. In cleaner regions low levels of soil N availability could limit the stimulation of growth normally caused by rising CO_2, but in areas of high N deposition the response to CO_2 may be increased. Warming, as already experienced since the 1980s, and in some upland areas most noticeable in winter, will have major effects on upland plant, animal and microbial life. There should be greater potential for lowland species to take to the higher ground but also increased threats for upland specialist organisms. However, the naturally acidic soils along with the unfavourable pollution climate - high rates of acid and N deposition, polluted cloud water and elevated exposure to ozone - in upland moorlands will likely affect competition between species, but in ways we cannot yet predict.

Regular management is probably the key to maintenance of good heather moorland condition in polluted areas of the country; experimental work indicates that N inputs advance the development of heather so that burning or cutting needs to be practised more often. However, overgrazing by sheep or deer should also be prevented; Mitchell and Hartley (2005) added N to experimental moorland plots in the Cairngorms and found that heather suffered significant decline, accompanied by an increase in grasses, only when grazing was allowed.

UK actions being taken to help reduce air pollution

Protocols under the UNECE Convention on Long Range Transboundary Air Pollution (CLRTAP), have already led to substantial emissions reductions for sulphur dioxide and nitrogen oxides. As a result, acid deposition in the UK has declined by approximately 50% over the past 12 years, mainly due to reductions in sulphur emissions. Under the latest CLRTAP agreement (the Protocol to Abate Acidification, Eutrophication and Ground-level Ozone) UNECE parties have agreed more stringent emission ceilings for SO_2 and NO_x as well as the first emission ceilings for NH_3, to be met from 2010. A major driver for agreement of these ceilings was the aim to reduce exceedance of critical loads for acidification and eutrophication across Europe. Critical loads are defined as the amount of acidity or nutrient N deposited on an ecosystem that, if exceeded, could lead to damage of that ecosystem. Critical loads are improved and refined as new data on ecosystem impacts become available. A recent update of UK critical loads has been undertaken and the report is available at: www.critloads.ceh.ac.uk .

Further Information

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ANNEX 5

Mini syntheses of impacts of acid and nitrogen deposition on montane heath

The impacts of acid and nitrogen deposition on:

Montane Heath

Montane heathlands comprise a range of dwarf-shrub, moss and lichen dominated communities occurring above the potential tree-line. They represent the most extensive remaining areas of near-natural habitat in the UK and are highly valued for the unique range of arctic and alpine species which they support. Montane areas form the headwaters of many river systems and play an important role in the regulation of water supply and quality for downstream uses such as drinking water and fisheries.

Montane moss and dwarf-shrub heaths are most common in Scotland, but have a restricted distribution south of the Scottish Highlands. They are thought to have declined over the last 50 years. There is also evidence of a loss of characteristic mosses and lichens and an increase in grasses. This is thought to be a result of past and current overgrazing combined with the effects of nitrogen (N) deposition and, increasingly, climate change (Thompson & Brown 1992).

The distribution of inputs of acidity and nitrogen across the UK

Energy production through the combustion of fossil fuels results in the emission of nitrogen oxides (NO_x) and sulphur dioxide (SO_2) into the atmosphere. Food production also causes pollutant emissions: ammonia (NH_3) from farm animal units and both ammonia (NH_3) and nitrous oxide (N_2O) from intensive fertiliser use. These pollutants are transported in the atmosphere affecting air quality and rainfall chemistry across the UK. This has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to N enrichment of semi-natural areas. Reductions in emissions due to policy control measures have resulted in lower quantities of sulphur and nitrogen oxides falling on different habitats but, due to increases in emissions from shipping, recovery has not been as fast as hoped for. Ammonia emissions increased sharply from the 1950s to 2000 and currently remain at these peak levels.

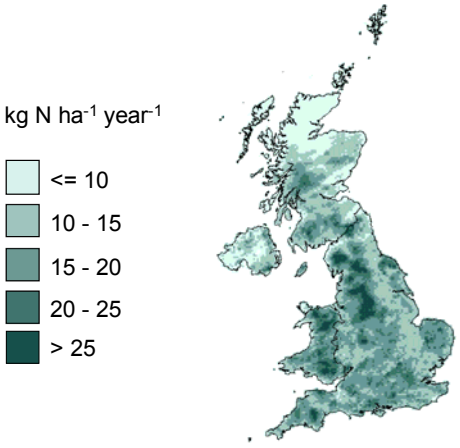

Fig 1a. Nitrogen (NOx + NHx) deposition measured 2003-2005

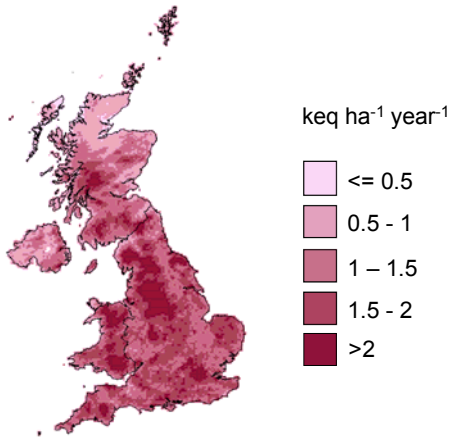

Fig 1b. Total acid deposition (S + NOx + NHx) measured 2003-2005

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Evidence of acidification and N-enrichment effects at the national scale

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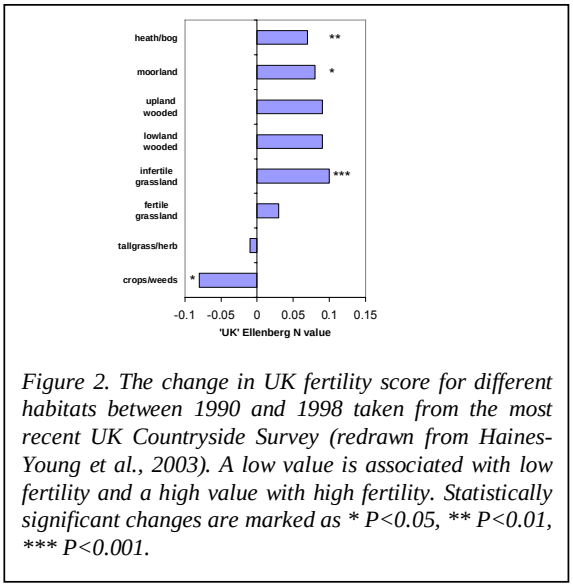

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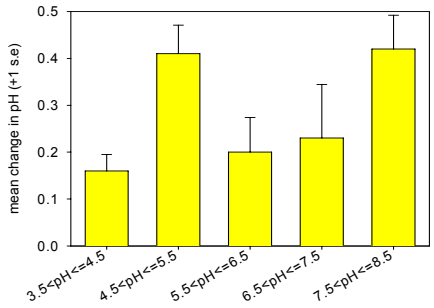

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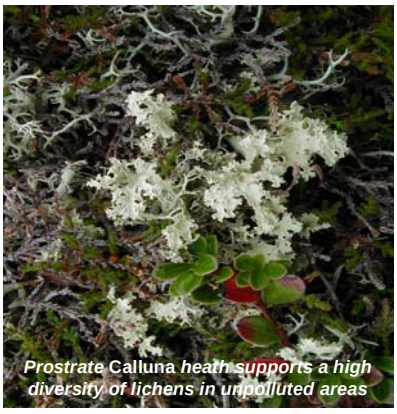
Current evidence for air pollution effects in montane heaths

The few studies which have been carried out in montane heathland ecosystems have focussed on two distinct communities: prostrate *Calluna vulgaris-Cladonia arbuscula* heath and *Carex bigelowii-Racomitrium lanuginosum* moss heath. The former is dominated by a carpet of prostrate dwarf shrubs with a diverse lichen community, while the latter is dominated by the montane moss *Racomitrium lanuginosum* (woolly hair moss) along with grasses and sedges. Both are climatically-determined 'climax communities' occupying large areas on exposed mountain summits and ridges in the UK.

South of the Scottish Highlands these communities are absent from many areas they occupied in the 1950s. Current summit vegetation in the southern Uplands, northern England and North Wales often consists of impoverished variants of these communities, with a reduced cover of mosses, lichens and dwarf shrubs, and a high prevalence of grasses. This degradation is thought to result from a combination of increasing N deposition (up to 56 kg N ha^{-1} yr^{-1} in montane areas of the UK) and high grazing pressures, both of which occurred over the period 1950-present. Separating the effects of these two factors is not easy and interactions between the two may substantially increase their impacts. Degradation of the ecosystem is not limited to effects on the plant community composition; studies on the Carneddau plateau in north Wales (Britton *et al.* 2005) have also shown reduced soil C:N ratios, loss of soil carbon from the most degraded areas and a high N content of both soils and plant tissues, compared with 'clean' sites in northern Scotland.

Degraded Racomitrium *heath in North Wales*

Both *Racomitrium* and *Calluna* have been shown to respond to N deposition by accumulating N in their tissues. Below 10 kg N ha^{-1} yr^{-1}, *Racomitrium* is able to use additional N supplies for growth but, as N deposition increases, excess N accumulates in the tissues. At very high levels of N deposition, the cell membranes of the moss become damaged, making them 'leaky', resulting in decreased growth and eventually shoot death (*Pearce et al*. 2003). The result is a reduced cover and depth of the moss mat and increased N availability in the soil as nutrients are released from decaying tissue. Reduction in moss cover may lead to soil erosion or allow expansion of grasses and sedges able to use the increased N supply. This in turn may decrease light availability to the moss, further reducing growth. Loss of the moss carpet and the thick organic layer below, which acts as both a sponge and a filter, may also have important consequences for hydrology and water quality downstream.



Prostrate Calluna *heath supports a high diversity of lichens in unpolluted areas*

In prostrate dwarf-shrub heaths in the UK and Scandinavia, N deposition has been shown to affect several aspects of community structure and function (Fremstad *et al.*, 2005). Nitrogen inputs as low as 10 kg N ha^{-1} yr^{-1} cause a reduction in the cover and species richness of the diverse lichen community associated with this habitat. *Calluna*, however, responds by increasing its shoot growth as long as phosphorus is not limiting. This community does not show the dramatic shift to grass dominance seen in lower altitude heaths, probably because of the limited amount of grasses present. Nitrogen addition also results in soil acidification and the associated loss of essential plant nutrients. Increased acidity mobilises toxic ions such as aluminium and heavy metals which are poisonous to terrestrial and aquatic life alike.

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How will climate change and management affect the impacts of air pollution?

The effects of air pollution on vegetation, soils and waters are not independent of other types of environmental change. Factors such as climate change and management may influence the sensitivity of ecosystems to air pollution. In some cases these will reduce damage while in other cases they may act to amplify effects. In montane heathlands two issues have been highlighted as being of major concern. These are the impact of domestic and wild herbivore populations and that of climate change. Studies in *Racomitrium* heath (van der Wal *et al.* 2003) have shown how grazing can enhance the negative effects of N deposition through trampling of the moss and deposition of dung, favouring grass growth. This interaction may be responsible for the degraded state of montane heaths south of the Highlands. Predicted future increases in drought frequency may also amplify N impacts. Montane habitats are exposed to extreme climates including the drying effects of high wind speeds during both summer and winter, and N deposition has been shown to increase drought sensitivity of key species. When also exposed to high levels of N deposition this can lead to increased winter injury in *Calluna* and reduced growth in *Racomitrium* (Jones *et al.* 2002).

Winter injury in *Calluna vulgaris* subject to high N load

UK actions being taken to help reduce air pollution

Protocols under the UNECE Convention on Long Range Transboundary Air Pollution (CLRTAP), have already led to substantial emissions reductions for sulphur dioxide and nitrogen oxides. As a result, acid deposition in the UK has declined by approximately 50% over the past 12 years, mainly due to reductions in sulphur emissions. Under the latest CLRTAP agreement (the Protocol to Abate Acidification, Eutrophication and Ground-level Ozone) UNECE parties have agreed more stringent emission ceilings for SO_2 and NO_x as well as the first emission ceilings for NH_3, to be met from 2010. A major driver for agreement of these ceilings was the aim to reduce exceedance of critical loads for acidification and eutrophication across Europe. Critical loads are defined as the amount of acidity or nutrient N deposited on an ecosystem that, if exceeded, could lead to damage of that ecosystem. Critical loads are improved and refined as new data on ecosystem impacts become available. A recent update of UK critical loads has been undertaken and the report is available at: www.critloads.ceh.ac.uk .

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ANNEX 6

Mini syntheses of impacts of nitrogen deposition on acid and calcareous grasslands

The impacts of nitrogen deposition on:

Acid and Calcareous Grasslands

Grasslands are the most extensive semi-natural plant communities in the UK countryside, with over 20% of our native species associated with these habitats (Preston *et al*., 2002). Unfertilised acidic and calcareous grasslands contain over 300 of the 540 grassland-associated native plant species. These two types of grassland have exceptional conservation and amenity value as a result of their floristic diversity. Unprecedented decreases occurred in the area of semi-natural grassland communities in the UK from 1930-1988, largely associated with agricultural expansion and intensification.

Of the 179 native species whose UK distribution declined in this period, nearly 40% are species of calcareous, unimproved or acidic grassland/heathland (Rich & Woodruff, 1996). Recent evidence indicates that the rapid increase in nitrogen (N) deposition that occurred in the latter half of the 20th century has been a major contributor to UK grassland biodiversity loss (Stevens *et al.*, 2004).

The long-term cumulative nitrogen deposition on UK grasslands

Energy production through the combustion of fossil fuels results in the emission of nitrogen oxides (NO_x) and sulphur dioxide (SO_2) into the atmosphere. Food production also causes pollutant emissions: ammonia (NH_3) from farm animal units and both ammonia (NH_3) and nitrous oxide (N_2O) from intensive fertiliser use. These pollutants are transported in the atmosphere affecting air quality and rainfall chemistry across the UK. This has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to N enrichment of semi-natural areas. Reductions in emissions due to policy control measures have resulted in lower quantities of sulphur and nitrogen oxides falling on different habitats but, due to increases in emissions from shipping, recovery has not been as fast as hoped for. Ammonia emissions increased sharply from the 1950s to 2000 and currently remain at these peak levels.

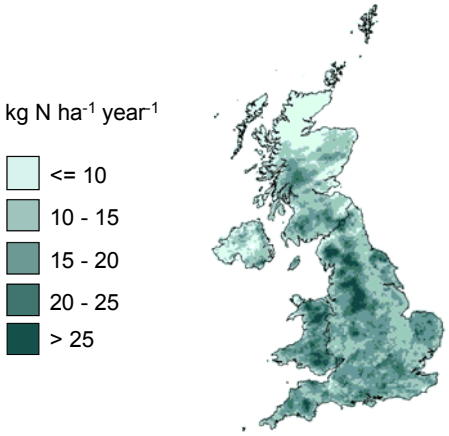

Fig 1a. Nitrogen (NOx + NHx) deposition measured 2003-2005

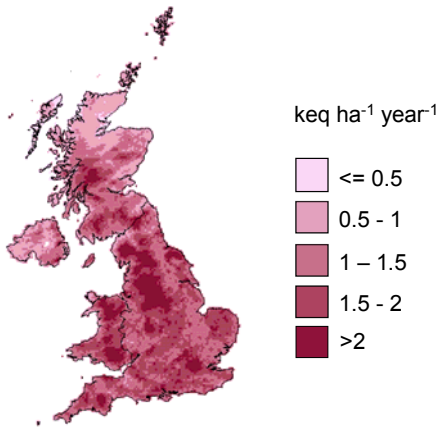

Fig 1b. Total acid deposition (S + NOx + NHx) measured 2003-2005

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Evidence of acidification and N-enrichment effects at the national scale

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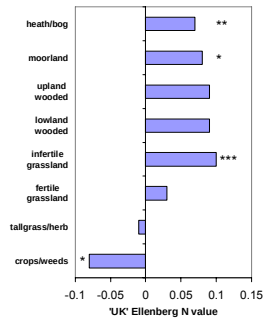

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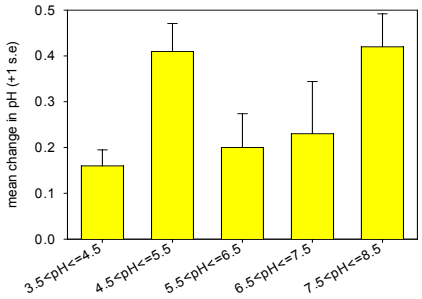

Figure 3. The increase in soil pH (0-15cm) from 1978-1998/9 at 767 sites across Great Britain (NEGTAP 2001)

Acidification of soils and waters recorded in some areas during the 20th century are now being reversed, reflecting the success of emission policies to reduce levels of acid deposition in the environment (e.g. Figure 3). There are still areas at risk, however, due to increases in sulphur emissions from shipping.

Why does air pollution affect our grassland, soil and waters?

The richness and diversity of species in unfertilised grasslands is dependent upon the large numbers of plants coexisting, without any single species being so dominant as to exclude others. Species-rich grasslands are typically associated with low productivity, generally as a result of nutrient limitation. Deposition of N can upset the delicate balance of competition between species, with increased nutrient availability causing eutrophication. These effects can be both direct - where some species such as mosses and lichens are damaged by acid rain, or indirect - where some species are out-competed by others (Carroll *et al.,* 2003). In addition, excessive N supply relative to plant demand causes N to be washed out of soil rather than taken up into plants, and this can lead to pollution of rivers and lakes. The acidifying components of N deposition also change soil chemistry and can have adverse effects on soil microbial activities and upon many plant species. The major changes in soil chemistry caused by N deposition include loss of calcium and other base cations in association with nitrate leaching, and increased solubility of aluminium. In the uplands, where the soils are often already highly leached due to the high annual rainfall, further base cation depletion and soil acidification are likely to have serious long-term impacts on the plant communities.

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Current evidence for nitrogen pollution effects on grasslands

Direct impacts of N enrichment on grassland biodiversity have been established from long-term N addition experiments in the Derbyshire Dales (Carroll *et al.*, 2003), confirming loss of species richness typically through the increase of grasses at the expense of forbs and mosses. These changes are progressive with time and are greater with higher rates of N deposition. One of the most sensitive indicators of N pollution impacts is the flowering of forbs in both calcareous and acid grasslands to which simulated pollutant N deposition has been applied in monthly spray additions for over a decade (Figure 4).

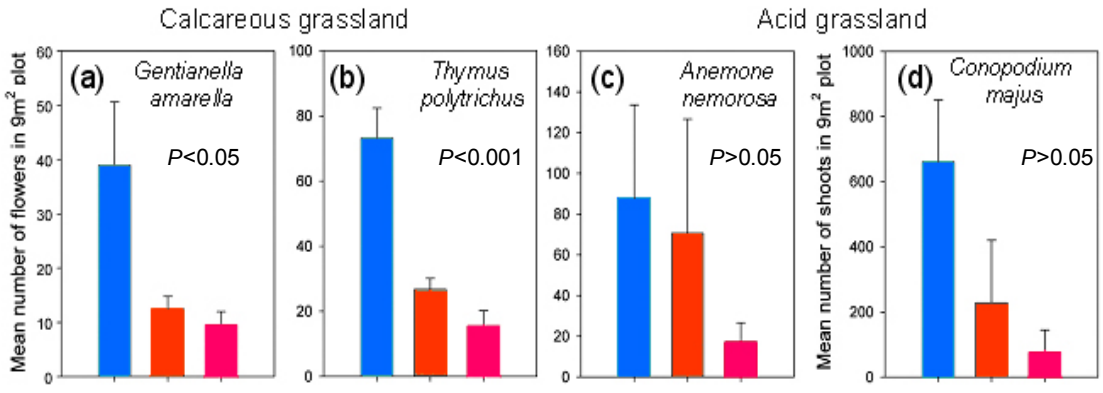

Figure 4. Numbers of flowers (a-c) and numbers of shoots (d) of plants in control plots ( ) and plots receiving simulated pollutant N deposition at 35 ( ) and 140 ( ) Kg N ha^{-1} year^{-1} for 10-11 years. (O'Sullivan & Leake, UKREATE).

Loss of flowering seriously compromises the amenity value of these grasslands, and is likely to have long-term effects on the sustainability of those species that depend upon seed production for regeneration, such as the biennial *Gentianella amarella*. Our findings that some species are particularly sensitive to N deposition is supported by the national monitoring that has also found declines in the extent of the important calcareous grassland species *Thymus polytrichus*, and *Gentianella campestris.* However, some of the change and loss of species may have already occurred during the early part of the 20th century (Emmett, 2007), or be hidden by the long term nature of the changes and variability inherent in grasslands.

In another UKREATE acid grassland experiment in Mid Wales, an increase in cover of grass species at the expense of bilberry was found following regular additions of N where grazing pressure was reduced to encourage the return of a more diverse flora. In more heavily grazed paddocks, the effect of N addition was not apparent, indicating the dominant effect grazing has in some systems. In both experiments, acidification and loss of base cations was observed in soil water draining from the experimental plots. These effects will be long lasting without human intervention since replacement of bases from rainfall and weathering contribute small inputs compared to the losses. The extent to which upland grasslands can naturally recover from the long-term inputs of acidic pollution remains uncertain, and may require addition of bases such as ground limestone or dolomite to facilitate recovery of damaged sites.

How will climate change and management affect the impacts of air pollution?

Air pollution interacts with both management and climate change to affect the sustainability of our fragile and nationally important grassland habitats. Drought and frost exacerbate physiological stress on plants caused by N pollution. Warming and drought mobilises N, increasing the risk of N enrichment of the plant community and leaching of N and acidity to linked streams and rivers. Some grassland plants receiving high rates of N deposition have increased susceptibility to frost and drought damage and may suffer increased attack from pathogens and insect herbivores. Drought increases the risk of grassland fires with potentially devastating effects on the species-rich communities that are dominated by slow-growing plants, many of which spread clonally. Where N deposition inhibits flowering and seed production this may compromise grassland recovery from seed-banks after extreme events such as fire or drought that damage or kill the turf. Appropriate grazing management is of absolutely vital importance for the maintenance of grassland communities and to prevent invasion by scrub and trees. However, overgrazing by sheep has been a particular problem in much of the upland grasslands causing the sward to break down and leading to increased erosion and oxidation of soil organic matter. Where grazing is reduced to encourage the return of heathland species we have found evidence that N deposition can slow their return. The use of specific management tools such as mowing with removal of clippings to deplete N enriched grasslands may help to reduce the effects of N pollution but further work is needed to validate this approach.

UK actions being taken to help reduce air pollution

Protocols under the UNECE Convention on Long Range Transboundary Air Pollution (CLRTAP), have already led to substantial emissions reductions for sulphur dioxide and nitrogen oxides. As a result, acid deposition in the UK has declined by approximately 50% over the past 12 years, mainly due to reductions in sulphur emissions. Under the latest CLRTAP agreement (the Protocol to Abate Acidification, Eutrophication and Ground-level Ozone) UNECE parties have agreed more stringent emission ceilings for SO_2 and NO_x as well as the first emission ceilings for NH_3, to be met from 2010. A major driver for agreement of these ceilings was the aim to reduce exceedance of critical loads for acidification and eutrophication across Europe. Critical loads are defined as the amount of acidity or nutrient N deposited on an ecosystem that, if exceeded, could lead to damage of that ecosystem. Critical loads are improved and refined as new data on ecosystem impacts become available. A recent update of UK critical loads has been undertaken and the report is available at: www.critloads.ceh.ac.uk .

Further Information

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ANNEX 7

Mini Syntheses of impacts of acid nitrogen deposition on sand dune habitats

The impacts of acid and nitrogen deposition on:

Sand dune habitats

Sand dune habitats are one of the most natural remaining vegetation types in the UK. They support over 70 nationally rare or red-data book species, and are a refuge for many species lost due to agricultural improvement of other lowland habitats. The open dune habitats in particular are important for a range of species: plants which are intolerant of competition; insects which require some bare soil for burrowing; and for threatened reptiles and amphibians such as the sand lizard, natterjack toad and great-crested newt.

Sand dunes are sensitive to many pressures, including: habitat loss, sea-level rise, climate change, agricultural improvement, tourist pressure, lack of management, and over-stabilisation. Nitrogen (N) deposition is thought to be a major contributor to over-stabilisation and species decline in UK dune systems.

The distribution of inputs of acidity and nitrogen across the UK

Energy production through combustion of fossil fuels results in the emission of nitrogen oxides (NO_x) and sulphur dioxide (SO_2) into the atmosphere. Food production also emits pollutants: ammonia (NH_3) from farm animal units, and both ammonia (NH_3) and nitrous oxide (N_2O) from intensive fertiliser use. These are transported in the atmosphere affecting air quality and rainfall chemistry across the UK. This pollutant deposition (Figure 1) has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to N enrichment of semi-natural areas. Reductions in emissions due to policy control measures have resulted in lower quantities of sulphur and nitrogen oxides falling on different habitats but, due to increases in emissions from shipping, recovery has not been as fast as hoped for. Ammonia emissions increased sharply from the 1950s to 2000 and currently remain at these peak levels.

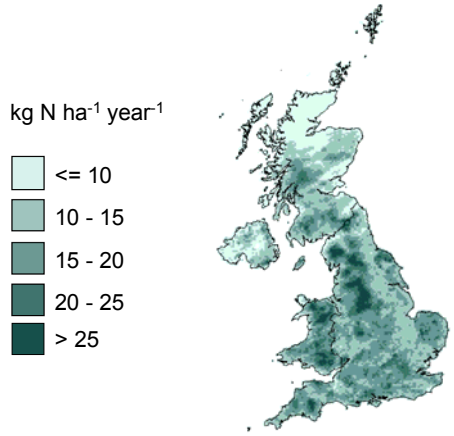

Fig 1a. Nitrogen (NOx + NHx) deposition measured 2003-2005

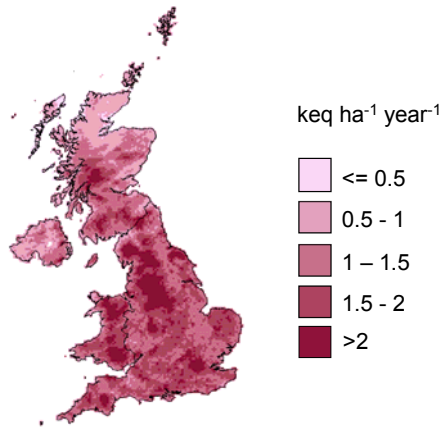

Fig 1b. Total acid deposition (S + NOx + NHx) measured 2003-2005

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Evidence of acidification and N-enrichment effects at the national scale

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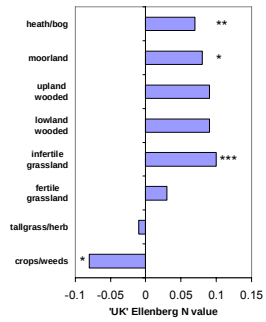

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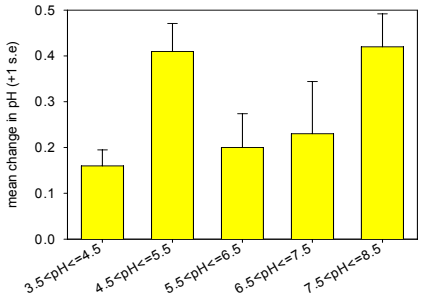

Figure 3. The increase in soil pH (0-15cm) from 1978-1998/9 at 767 sites across Great Britain (NEGTAP 2001)

Acidification of soils and waters recorded in some areas during the 20th century are now being reversed, reflecting the success of emission policies to reduce levels of acid deposition in the environment (e.g. Figure 3). There are still areas at risk, however, due to increases in sulphur emissions from shipping.

Why does air pollution affect our soils, vegetation and waters?

Although rainfall is naturally acidic, additional acidity either introduced directly by sulphur dioxide and nitrogen oxides or formed during the breakdown and uptake of ammonia has affected waters, soils and vegetation in the UK. The pH of lakes and rivers fell during the last century, in turn affecting populations of fish, invertebrates and water plant communities. Soils also became more acidic, affecting organic matter breakdown and soil nutrient balance. Soil acidification increases the solubility of some elements such as aluminium in the soil solution, which can be toxic to plant roots at high concentrations. Pollutants are also deposited to vegetation directly as gases, aerosols and in fogs and mists, and can cause direct damage to plants at high concentrations.

Emissions of nitrogen oxides and ammonia can lead to N enrichment (eutrophication). These problems can result in a loss of biodiversity in sensitive ecosystems because N-loving species benefit at the expense of other species of conservation interest that contribute so much to the character of semi-natural habitats. This happens due to nutrient imbalances, increased susceptibility to climatic stress and higher levels of insect or fungal damage which all affect the balance of competition between species.

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Current evidence for air pollution effects in sand dunes

Although physical processes, hydrology and succession have been extensively studied in sand dunes, research into air pollution effects in sand dunes is limited. A review on the effects of N deposition on sand dune habitats concluded that they were likely to be at risk of eutrophication. The greatest impact is likely to be on the early successional communities which are important for many of the sand dune rarities.

A survey approach of UK sand dunes along a gradient of N deposition showed significant correlations of vegetation, soil and groundwater parameters with increasing N deposition (Jones *et al*. 2004). In semi-fixed (open) dune habitats, cover of marram grass (*Ammophila arenaria*) and total biomass increased. In fixed dune grasslands, plant species diversity decreased (Figure 4) and biomass increased. Soil parameters showed surprisingly that the C:N ratio increased and available N decreased. Dissolved organic nitrogen (DON) concentrations in groundwater also increased. Experimental evidence from sand dune mesocosms in the Netherlands has shown an increase in cover of grasses and sedges and a decline in herbaceous species with increased N deposition (van den Berg *et al*. 2005). As a result of UK and Dutch studies, the suggested critical load range for sand dunes is 10 – 20 kg N ha^{-1} yr^{-1} (Figure 5).

Figure 4. Species richness, fixed dune grassland

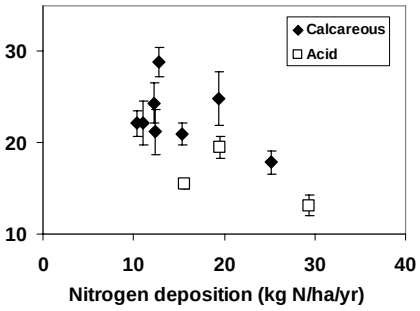
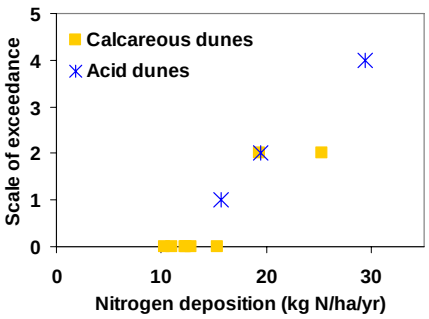


Figure 5. Critical load exceedance

Nitrogen retention in sand dune soils is poor, due to the low levels of organic matter. Nitrogen addition experiments in the Netherlands showed that leaching of N varied from 0 - 70 % of inputs (ten Harkel *et al.* 1998). A UK study under ambient N deposition, showed that total leaching losses of N varied from 15 – 65 % of inputs, mainly as DON. Losses were lowest in ungrazed vegetation on calcareous soils and were highest in rabbit grazed vegetation (Jones *et al.* 2005).

Although much N is lost through leaching, enough is retained to cause adverse effects. An N manipulation experiment at Newborough Warren in North Wales (Plassmann, 2006) has shown that even low additions of N (within the critical load range) result in significant accumulation of N within the moss layer, which will be released to the rest of the soil-plant system as those moss shoots die. Phosphorus limitation is common in many UK dunes and may reduce vegetation responses to excess N deposition. However, as soil pH approaches pH 5, phosphorus becomes more available to plants, thus increasing the likelihood of adverse impacts of N deposition in more acidic systems.



In UK dunes, the main impact of N is that of eutrophication, and acid dune systems appear to be more sensitive to N inputs than calcareous dunes (Figure 5). Soil acidification as a result of acid deposition has relatively little impact in UK dunes because sand dune soils are generally well-buffered, with the exception of the few acidic dune systems. However, in the Netherlands where the sand usually has a lower initial carbonate content, both acidification and eutrophication have resulted in a decline of rare pioneer species in dune slacks sensitive to acidification (Sival & Strijkstra-Kalk, 1999).

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How will climate change and management affect the impacts of air pollution?

Management can be used to mitigate the effects of N deposition. Both mowing and grazing tend to increase species diversity, and mowing (with removal of cuttings) removes N from the system. Managed grazing is frequently used to reduce the vigour of competitive species and to retain species of conservation interest. Natural grazers such as rabbits also play a key role, but need to be managed in conjunction with conventional grazing to allow for natural population fluctuations. Although the net removal of N by grazers is almost negligible, over an extended period of time rabbit grazing effectively reduces the soil N pool by slowing the rate of organic matter accumulation compared with ungrazed habitats. More extreme management (turf-stripping or topsoil inversion) can be used to remove nutrient-rich surface layers and re-create earlier successional habitats.

The effects of climate change and interactions with air pollution are uncertain. In many areas sea-level rise will result in loss of sand-dune habitat, and the corresponding rise in water table will increase the depth and extent of dune slacks. Early successional habitats, already impacted by N deposition, are most at risk. Climate change may increase or decrease mobility of dune systems, depending on the balance of rainfall, storminess, wind direction and wind speeds, and rising temperatures may affect the ranges of key species.

UK actions being taken to help reduce air pollution

Protocols under the UNECE Convention on Long Range Transboundary Air Pollution (CLRTAP), have already led to substantial emissions reductions for sulphur dioxide and nitrogen oxides. As a result, acid deposition in the UK has declined by approximately 50% over the past 12 years, mainly due to reductions in sulphur emissions. Under the latest CLRTAP agreement (the Protocol to Abate Acidification, Eutrophication and Ground-level Ozone) UNECE parties have agreed more stringent emission ceilings for SO_2 and NO_x as well as the first emission ceilings for NH_3, to be met from 2010. A major driver for agreement of these ceilings was the aim to reduce exceedance of critical loads for acidification and eutrophication across Europe. Critical loads are defined as the amount of acidity or nutrient N deposited on an ecosystem that, if exceeded, could lead to damage of that ecosystem. Critical loads are improved and refined as new data on ecosystem impacts become available. A recent update of UK critical loads has been undertaken and the report is available at: www.critloads.ceh.ac.uk .

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ANNEX 8

Mini Syntheses of impacts of sulphur
and nitrogen deposition on woodlands

The impacts of sulphur and nitrogen deposition on:

woodlands

Woodlands are valued for their high conservation status, contribution to the landscape, ability to store carbon and provision of public access and recreation opportunities. Recent surveys have shown no evidence of widespread damage from air pollution to forest trees, but excess nitrogen (N) deposition has been implicated in observed changes in the composition of woodland plant communities. Plant surveys indicate that N-demanding species such as bramble are becoming more abundant while others, such as bilberry, are declining. It is also thought that a lack of regeneration due to under-management in recent decades and browsing by deer has contributed to a degradation of ecological condition in some woodlands.

The distribution of inputs of acidity and nitrogen across the UK

Energy production through the combustion of fossil fuels results in the emission of nitrogen oxides (NO_x) and sulphur dioxide (SO_2) into the atmosphere. Food production also causes pollutant emissions: ammonia (NH_3) from farm animal units and both ammonia (NH_3) and nitrous oxide (N_2O) from intensive fertiliser use. These pollutants are transported in the atmosphere affecting air quality and rainfall chemistry across the UK. This has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to N enrichment of semi-natural areas. Reductions in emissions due to policy control measures have resulted in lower quantities of sulphur and nitrogen oxides falling on different habitats but, due to increases in emissions from shipping, recovery has not been as fast as hoped for. Ammonia emissions increased sharply from the 1950s to 2000 and currently remain at these peak levels.

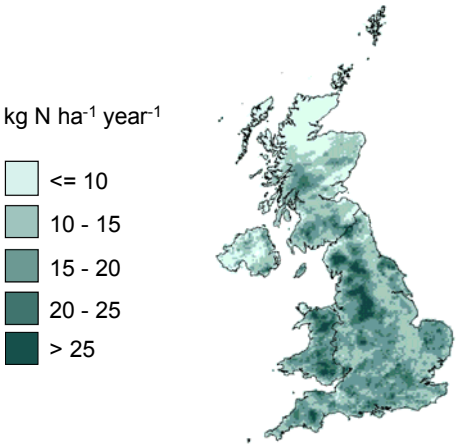

Fig 1a. Nitrogen (NOx + NHx) deposition measured 2003-2005

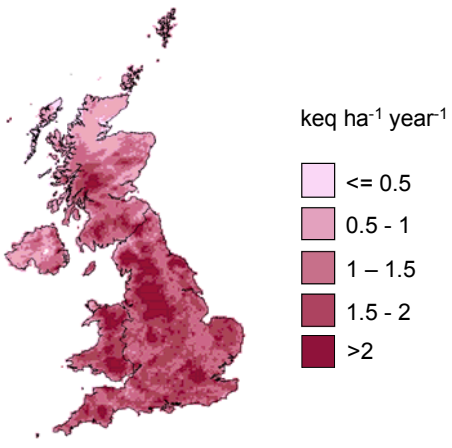

Fig 1b. Total acid deposition (S + NOx + NHx) measured 2003-2005

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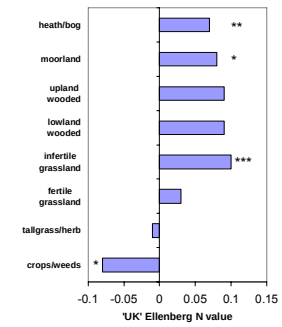

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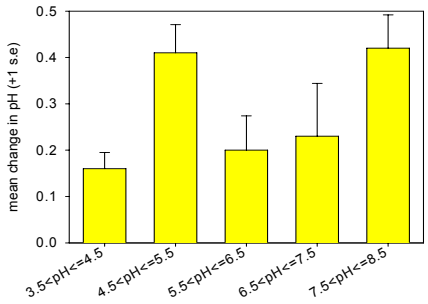

Figure 3. The increase in soil pH (0-15cm) from 1978-1998/9 at 767 sites across Great Britain (NEGTAP 2001)

Acidification of soils and waters recorded in some areas during the 20th century are now being reversed, reflecting the success of emission policies to reduce levels of acid deposition in the environment (e.g. Figure 3). There are still areas at risk, however, due to increases in sulphur emissions from shipping.

How does air pollution affect our waters, soils and vegetation?

Although rainfall is naturally acidic, additional acidity either introduced directly by sulphur dioxide and nitrogen oxides or formed during the breakdown and uptake of ammonia has affected waters, soils and vegetation in the UK. The pH of lakes and rivers fell during the last century, in turn affecting populations of fish, invertebrates and water plant communities. Soils also became more acidic, affecting organic matter breakdown and soil nutrient balance. Soil acidification increases the solubility of some elements such as aluminium in the soil solution, which can be toxic to plant roots at high concentrations. Pollutants are also deposited to vegetation directly as gases, aerosols and in fogs and mists, and can cause direct damage to plants at high concentrations.

Emissions of nitrogen oxides and ammonia can lead to N enrichment (eutrophication). These problems can result in a loss of biodiversity in sensitive ecosystems because N-loving species benefit at the expense of other species of conservation interest that contribute so much to the character of semi-natural habitats. This happens due to nutrient imbalances, increased susceptibility to climatic stress and higher levels of insect or fungal damage which all affect the balance of competition between species.

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Current evidence for air pollution effects in British woodland

Long-term monitoring indicates that there is no widespread damage to forests and trees in the UK as a result of atmospheric pollution. The network of 350 Forest Condition Survey Plots, on which diameter increment and crown condition have been measured since 1987, shows no link between soil acidification and tree vitality. However, soil function may be affected leading to more subtle effects. For example, the Forest Condition Survey has identified a link between N deposition and the level of insect damage to Scots pine (Figure 4). An in-depth analysis of the ground flora associated with beech woodlands across the network has also revealed a relationship between the distribution of N-demanding species and distance to woodland edge (Figure 5) - a clear signal of the effects of N deposition. This observation is supported by recent surveys of plant communities, including Countryside Survey 2000, the New Plant Atlas and a re-analysis of ecological condition of over 100 semi-natural woodlands first surveyed in1971 (Kirby *et al.*, 2005), as well as by studies of the effects of point sources of N pollution (Pitcairn *et al.*, 2002). However, there are caveats that should be applied when interpreting observed trends as a clear signal of the effects of N deposition. First, the level of woodland management has declined over the past two to three decades as a result of the poor economic climate of the forestry sector, reducing light levels on the forest floor thus favouring shade-tolerant species. Second, the woodlands have aged, which similarly affects the composition of the ground flora (Figure 6).

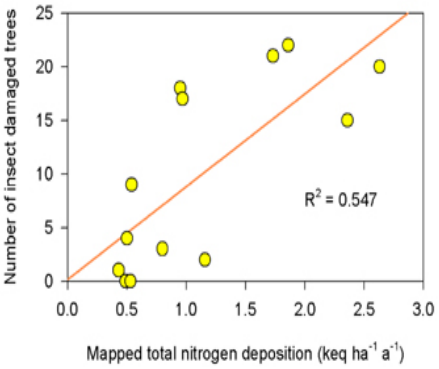
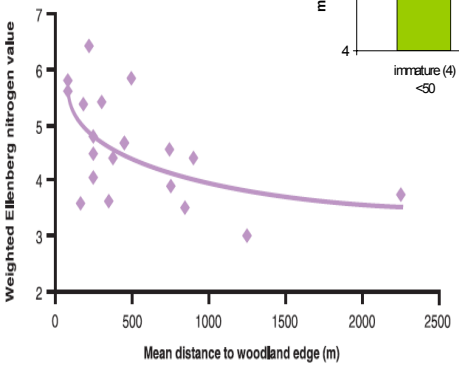


Figure 4 (above). Relationship between observed number of insect damaged trees and mapped total nitrogen deposition for Forest Condition Survey Scots pine plots (NEGTAP, 2001).

Figure 5 (below). Changes in the nitrogen demand of beech woodland ground flora with proximity to the woodland edge (Kennedy, 2003).



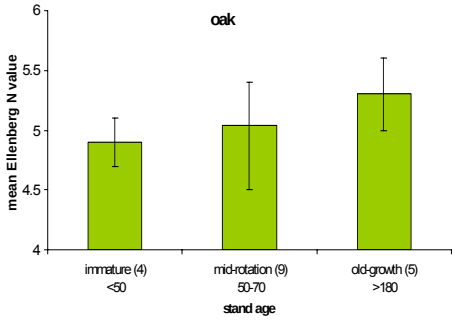


Figure 6 (above). Variation in nitrogen demand of ground flora of oak woodland with stand age.

Soil solution chemistry from the Intensive Forest Monitoring Network has indicated some recovery over the past ten years from high historical pollution loading, as a result of emission control policies. The most dramatic observation is the downward trend in tree foliar sulphur concentrations, corresponding to a reduction in soil solution sulphate concentrations across the network. Indeed, it is likely that sulphur deficiency may become an issue in the near future in some regions. There is little evidence to indicate a reduction in N deposition or its impacts; on the contrary, there is continuing evidence of the effects of high N deposition, particularly in areas dominated by intensive agriculture, such as East Anglia. At one site in Thetford forest, where N deposition in throughfall has been measured as 25 kg ha^{-1} yr^{-1}, soil solution pH has fallen by 2 units since monitoring began in 1995 and nitrate concentrations in soil solution as high as 150 mg l^{-1} have been measured. Nettle, a species with high N demand, has also increased its dominance of the ground flora over this period (Vanguelova *et al.*, 2007).

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How will climate change and management affect the impacts of air pollution?

The predicted changes to the climate of the UK are likely to have significant impacts on woodland plant communities. Limited evidence suggests that rising CO_2 levels may favour weedy species, at the expense of slower growing species, possibly compounding the impacts of N deposition. Where water is not limiting, the combined effects of N deposition and rising CO_2 levels are likely to lead to significant increases in forest growth. There is already some evidence for this in old-growth oak in southern England (Broadmeadow, 2004). Climate change will also affect soil processes, leading to interactions between growth, nutrient cycling and pollutant inputs. For example, mineralisation rates will increase thus affecting nutrient availability, while nutrient uptake and leaching may be affected by changing rainfall patterns and a longer growing season (Bradley *et al.*, 2005).

Conventional forest management increases the input of N that a forest ecosystem can withstand, by removing a proportion in timber. At the same time, base cations are removed reducing the ability of the ecosystem to withstand acid deposition, particularly woodlands established on soils low in base cations. The future level of woodland management, particularly in response to an increased utilisation of woodfuel for climate change mitigation objectives, will affect how woodlands respond to the continuing effects of air pollution and it will be important to ensure that this interaction is accommodated in developing new climate change policy and forest management practices.

UK actions being taken to help reduce air pollution

Protocols under the UNECE Convention on Long Range Transboundary Air Pollution (CLRTAP), have already led to substantial emissions reductions for sulphur dioxide and nitrogen oxides. As a result, acid deposition in the UK has declined by approximately 50% over the past 12 years, mainly due to reductions in sulphur emissions. Under the latest CLRTAP agreement (the Protocol to Abate Acidification, Eutrophication and Ground-level Ozone) UNECE parties have agreed more stringent emission ceilings for SO_2 and NO_x as well as the first emission ceilings for NH_3, to be met from 2010. A major driver for agreement of these ceilings was the aim to reduce exceedance of critical loads for acidification and eutrophication across Europe. Critical loads are defined as the amount of acidity or nutrient N deposited on an ecosystem that, if exceeded, could lead to damage of that ecosystem. Critical loads are improved and refined as new data on ecosystem impacts become available. A recent update of UK critical loads has been undertaken and the report is available at: www.critloads.ceh.ac.uk .

Further Information

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